1
2 **Pollinator habitat plantings benefit wild, native bees, but do not necessarily favor rare species**

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ABSTRACT

1. Installing pollinator habitat is a ubiquitous conservation tool, but little is known about which pollinator taxa require support, or which benefit from habitat installations.

2. We studied the response of rare and common bees to pollinator habitat enhancement. We used independent regional datasets to designate bee species as common or rare based on their rank according to one of three metrics: a) site occurrence frequency, b) local relative abundance, and c) geographic range size. We asked whether the abundance or richness of rare and common bees were greater in pollinator habitat, relative to old-field controls. Because we used an arbitrary, quantile-based cutoff to categorize species rarity, we conducted sensitivity analysis and controlled for rarity classification errors with a null model. With this null model, we determined whether rare and/or common species responded to pollinator habitat disproportionately, compared to the expectation for 'typical' bee species.

3. We found that the number of individuals and of species designated as rare based on local relative abundance was greater in pollinator habitat enhancements. The number of individuals from bee species designated as rare based on site occurrence was lower in pollinator habitat enhancements, but the number of species was not clearly different between habitat types. We did not find a clear positive nor negative effect of habitat enhancement for species designated rare based on geographic range size. For all three rarity metrics, common species increased in abundance and richness in pollinator habitat relative to controls. Null models indicated that in most cases, neither rare nor common species disproportionately benefited from pollinator habitat.

4. *Synthesis and Applications:* Our results suggest that pollinator habitat can lead to an increase in the abundance and richness of bees, including species that are rare and that are common. However, rare species appeared to respond differently than typical species, and depending on how species were classified as rare, could display muted or even negative responses to habitat enhancement. Targeting rare species with specific floral resources or unique habitat types may lead to better outcomes for rare and threatened species.

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1 | INTRODUCTION

Habitat restoration is increasingly an important tool for conserving biodiversity (Hobbs 2007; Brudvig 2017). Currently, considerable government funding goes into restoring habitat for pollinators (Garbach & Long 2017). The primary method by which land managers restore habitat is the planting of wildflowers to provide resources such as pollen and nectar (Tonietto & Larkin 2018). For wild, native bee communities, the lack of floral resources is likely one of the main limiting factors (Roulston & Goodell 2011) and has been linked to declines in native bee health and abundance as well as species extinction (Carvell *et al.* 2006; Williams *et al.* 2012; Ollerton *et al.* 2014; Crone & Williams 2016). Increasing floral cover and diversity increases the overall diversity and abundance of bee pollinators (Blaauw & Isaacs 2014; Kleijn *et al.* 2015; Scheper *et al.* 2015; Williams *et al.* 2015; Tonietto & Larkin 2018; Albrecht *et al.* 2020).

Rare species comprise the bulk of biodiversity (McGill *et al.* 2007), and are often at greater risk of extirpation (Lande 1988; Manne *et al.* 1999; Payne & Finnegan 2007; Harnik *et al.* 2012). Conservation practices are often implemented to protect or benefit rare species *per se* (Hallett *et al.* 2013). However, few studies have specifically examined whether particular groups of pollinator species benefit from pollinator plantings (but see Pywell *et al.* 2012; Scheper *et al.* 2015; Sutter *et al.* 2017). There are reasons to expect that pollinator plantings both would, and would not, benefit rare species. On the one hand, such plantings could have a disproportionately positive effect on rare species if they contained specific resources that are limiting to those rare species in the wider landscape (Gibbs *et al.* 2008; Swarts & Dixon 2009). On the other hand, pollinator plantings in a degraded landscape might benefit only species that persist in these landscapes, and are thus resilient to anthropogenic change and likely not of greatest conservation concern (Kleijn *et al.* 2015). Lastly, restored pollinator plantings may similarly benefit both rare and common species. In this case, changing plant seed mixes to increase flower species preferred by rare bees may lead to better conservation outcomes for all species (Sutter *et al.* 2017; MacLeod *et al.* 2020).

An important first step in assessing the impact of habitats on rare species is to define which species are, in fact, rare. Rarity can be defined using multiple metrics, including local abundance, site occurrence frequency across a larger area, and/or geographic range size (Rabinowitz 1981; Gaston 1994). Species categorized as rare based on different metrics may be sensitive to different ecological processes. Species with low local abundance are prone to local extinction via stochastic demographic processes (Lande 1988). Site occurrence frequency can predict extinction risk (Payne & Finnegan 2007; Harnik *et al.* 2012) because species that are found at fewer sites are less likely to recolonize following disturbance (Fagan *et al.* 2002). Further, species that are only at a few sites may be habitat specialists and at risk if those particular habitats are lost (Rabinowitz 1981). A third metric, geographic range size, is an important predictor of extinction in the fossil record (Kiessling & Aberhan 2007; Payne *et al.* 2011). Species with small ranges may be especially worthy of conservation action because populations may be less able to escape more local- or regional-scale disturbances such land-use change (Manne *et al.* 1999; Manne & Pimm 2001).

Here, we test whether pollinator habitat installations, created through wildflower plantings developed for pollinators (herein pollinator planting) generally, benefit rare and/or common species of native bees. We sampled native bees in a paired study design, in which each pollinator planting 'treatment' was compared to a nearby semi-natural old field 'control'. Old

92   fields provide a realistic control habitat type because in our study region, they are the early
93   successional habitat that regenerates without any intervention. They also provide a conservative
94   control for assessing the effect of pollinator plantings, because they are one of the best habitats
95   for wild bees in the region of this study (Mandelik *et al.* 2012). To compare bee communities
96   between pollinator plantings and controls, we collected all species of wild, native bees at 16 pairs
97   of sites over five years. Because there are no comprehensive assessments (such as Red Lists) of
98   bee rarity for North America, we categorized species as regionally rare or common based on
99   large, independent datasets. We asked two questions: 1) Do pollinator habitat plantings benefit
100   rare and/or common bees? 2) Do pollinator habitat plantings benefits accrue disproportionately
101   to either rare or common bee species? We answered these questions categorizing species as rare
102   or common based on one of three metrics: a) local relative abundance, b) site occurrence
103   frequency, and c) geographic range size

104

2 | METHODS

2.1 Study Design and Data Collection.

107   This study took place from 2011 to 2015 in New Jersey and eastern Pennsylvania, USA
108   (Fig. 1). We selected a total of 16 pollinator plantings that were separated by at least 4.5km. All
109   habitats were installed by private landowners following United States Department of Agriculture
110   National Resource Conservation Service guidelines for pollinator habitat. We used a paired
111   design and at each pollinator planting chose a nearby old field as a control plot. Old-field control
112   plots were located between 200-800m from the pollinator planting, which allowed us to
113   minimize the spatial variation in native bee communities and landscape context between the
114   habitat and controls. Hereafter, we refer to a pollinator planting and its paired old-field control
115   together as a site.

116   To determine bee use of pollinator planting and old-field controls, we sampled bees along
117   40 m transects. In any given site- by- year combination, we sampled either 2 or 4 transects,
118   always with the same number of transects and equal effort at both the treatment and control plots
119   within a site. We walked along each transect and collected all wild, native bees visiting flowers
120   within 1m of each side of the transect. Transects were sampled for 10 minutes each, with the
121   timer stopped to process specimens. Each transect was sampled once in the morning and once in
122   the afternoon. To minimize variation due to weather, we sampled bees in dry, warm (>14C), and
123   still conditions (winds below 3.5 m/s). To minimize observer bias, each collector completed the
124   same number of transects in the pollinator restoration and old-field control each day. Sites were
125   sampled approximately once a month from June through September for a total of four rounds per
126   year. Sites were sampled for 1-4 years. This resulted in a total of over 293 hours of net
127   collecting, not including time to process specimens.

128   All bees were pinned, labeled, and identified to species or species complex (Table S1).
129   To identify bees to species, we used the following taxonomic revisions (Mitchell 1960; LaBerge
130   1961; Mitchell 1962; LaBerge 1967; Ribble 1968; LaBerge 1971, 1973; LaBerge and Ribble
131   1975; LaBerge 1977; Bouseman and LaBerge 1979; LaBerge 1980, 1986; McGinley 1986;
132   LaBerge 1987; Laverty and Harder 1988; LaBerge 1989; Coelho 2004; Gibbs 2011; Rehan and
133   Sheffield 2011; Gibbs et al. 2013) and keys available online (Arduser 2016; Larkin et al. 2016).
134   With the exception of the three most common bumble bee species (*Bombus impatiens, B.*

135 *bimaculatus,* and *B. griseocollis*), and the carpenter bee *Xylocopa virginica*, for which only
136 voucher specimens were kept, all specimens are stored in the Winfree Lab at Rutgers University
137 in New Brunswick, NJ USA. This dataset is hereafter referred to as the pollinator planting
138 dataset and is distinct from the independent datasets used for determining rarity, described
139 below.

140

141 2.2 Determining Rarity

142   Using independent datasets, we categorized species as rare or common based on three
143 metrics: local relative abundance, site occurrence frequency, and geographic range size. Species
144 were rare or common depending on whether they were in the lowest or highest quartile (Gaston
145 1994). Descriptions of the datasets and which datasets were used for each rarity metric follow
146 below.

147

148 2.2.1 | Local Relative Abundance

149   To determine whether a species was rare or common in terms of local relative abundance
150 and site occurrence, we compiled a large dataset of bee specimens collected in the states of New
151 Jersey, Pennsylvania, and New York, USA. The specimens came from six studies, conducted by
152 members of the Winfree research group, in which bees were collected using pan traps or hand
153 nets between 2003 and 2016 (Table S2). In all of these studies, collectors did not target particular
154 species or species groups, and collection took place irrespective of abundance; therefore, we
155 assumed that specimen records from these datasets, on average, reflect actual bee abundances at
156 sites in our region, and use these records to compute relative local abundance within sites.

157   We filtered records to generate rarity designations for our study region using the
158 following criteria. First, to evaluate rarity within the same region as the pollinator plantings, we
159 drew a rectangle where each edge was 100km from the nearest site in each cardinal direction.
160 Second, we filtered records for phenology, and used only records collected within two weeks of
161 the year from the start and end of the collection events of our restoration study. This could mean
162 that in our filtered dataset, bee species that are common either outside of our spatial scale or in a
163 different season may be classified as rare. Third, we removed exotic species (1.4% of all
164 individual records and <5% of species across the 6 reference data sets). Our study focused
165 exclusively on native species for a number of reasons. First, both the abundance and site
166 occurrence frequencies of exotic species are likely driven by unique factors such as human-
167 mediated colonization events and time since invasion. Second, as many of our specimens from
168 the independent datasets come from historic collections, exotic species are likely
169 underrepresented. Finally, we were most interested in characterizing native species; exotic
170 species are rarely the target of conservation efforts.

171   We then calculated local relative abundance as follows: First, to avoid artifacts related to
172 small sample sizes, we removed all records from sites at which fewer than 10 total specimens
173 had been collected. Second, we calculated relative abundance for each species at each location.
174 Abundances for each species at each locality were summed across years and sample rounds.
175 Third, for each species, we calculated mean local relative abundance across sites by summing the
176 relative abundances at all sites at which a species was present, and dividing this sum by the
177 number of sites at which that species was detected. Thus, when a species was not recorded at a

178 site, that site was not included in the local relative abundance calculation for that species; this
179 exclusion of zeros was done to make our measure of relative abundance distinct from site
180 occupancy (below). Finally, there were 10 species (represented by 47 specimens) that were
181 collected in the pollinator plantings dataset but were not recorded at the sites used to calculate
182 relative abundance. These species were removed from the analysis for local relative abundance.
183 In total, the data used to calculate local abundance consisted of 52,885 specimens and 294
184 species. Collections took place at 261 different locations (Fig. 1).

185 **2.2.2 | Site Occurrence Frequency**

186 To assess rarity based on site occurrence frequency, we used the same dataset used for local
187 relative abundance (see above), supplemented by a previously published dataset compiled from
188 species records from nine museums (Bartomeus *et al.* 2013). This dataset includes over 11,000
189 collection events, spanning 139 years (1870-2010). The data set is filtered to use only one record
190 (individual bee) per species per collection event to reduce biases that might be present in
191 museum collections. Specifically, collectors were unlikely to collect each species proportional to
192 its relative abundance. Thus, we use this data set for the occupancy analysis but not the
193 abundance analysis (above). For use in the occurrence analysis, we standardized the abundance
194 dataset to presence-absence form also (i.e., retained only one individual of each species per site-
195 date). We filtered the museum dataset for phenology, spatial extent and native status in the same
196 manner as the abundance dataset. After filtering and combining, the occurrence data set included
197 59,918 individual records of 384 species across 992 different locations (Fig. 1). To determine
198 site occurrence frequency, we counted the number of sites at which each species was collected.
199 Finally, we collected five species (represented by 35 specimens) in the pollinator habitat dataset
200 but that were not recorded at all in the site occurrence dataset. We categorized these species as
201 rare based on site occupancy.

202

203 **2.2.3 | Geographic Range Size**

204 To assess geographic range size, we used the methods from a study completed by some
205 of the present coauthors (Harrison *et al.* 2017). This study used recorded state records from the
206 Discover Life AMNH_BEES database (Ascher & Pickering 2019). A minimum bounding
207 polygon was then calculated to contain the centroids of each state in which each species was
208 present. For details see (Harrison *et al.* 2017).

209

210 **2.3 | Quantile Designation of Rare and Common Species***:*

211 For *local relative abundance*, 74 species in the regional dataset were designated as rare
212 (local relative abundance = <0.81%) and 73 species as common (local relative abundance =
213 >3.7%). For *site occurrence frequency*, there was an unequal number of species in some quartiles
214 due to ties. Ties for the first and second quartiles and the third and fourth quartile were placed in
215 the first and fourth quartile respectively. There were 105 species designated as rare (< 4 sites)
216 and 94 as common (>26 sites). For *geographic range size* we designated 96 species as rare and
217 96 as common. A table of all species and their rarity classifications can be found in Table S1.

218

219 **2.4 Data Analysis**

2.4.1 | Do pollinator habitat restorations benefit rare and or common bees?

We designated each species collected in the pollinator restoration dataset as rare or common (or neither, constituting those between the 25th-75th percentiles) for each of the three rarity metrics. For each pollinator planting and old-field control at each site, we then summed (across all years and sampling rounds) the number of individuals (hereafter, "abundance") and species (hereafter, "richness") of rare and common bees. To determine how pollinator planting affected the richness or abundance of rare, and separately, common bees, we fit generalized linear models with either richness or abundance of either rare or common bees, according to one of the three metrics, as the response variable. As predictors in each model, we incorporated treatment (pollinator planting or old-field control) as a fixed effect. Additionally, because we used a paired study design, with one wildflower planting and one control at each site, we included site as a random effect. We fit generalized linear mixed-effects models for each response variable with a negative binomial error structure using the function *glmer_nb* from the R package "lme4" (Bates *et al.* 2015). Model assumptions were checked using the DHARMa package (Hartig 2020). This analysis was conducted using R 4.0.5 (R Core Team 2021).

2.4.2 | Do pollinator habitat restoration benefits accrue disproportionately to either rare or common bee species?

The models described above test for increases in the abundance or richness of a specified group of species (e.g. those that have smaller geographic ranges), regardless of how other species respond. If managers believe that rare species are especially vulnerable to global change, it may be strategic to focus management actions that benefit these species. We generated a null model to test whether the species we designated as rare, or those we designated as common, especially benefit from habitat enhancement, or whether they respond about as strongly as expected, given that the response group consists of about one quarter of the species in our study region.

In each iteration of our null model, we designate a random subset of species as rare or common, where the number of rare and common species in the null group is the same as the number of rare and common species in the empirical data. We used our observations of these species in the pollinator habitats and old-field controls, and refit the models for each rarity criterion (i.e. for each group of [null] "rare" or [null] "common" species) and each response (i.e. richness or abundance). We then compared the model estimates for the effect of habitat enhancement for the species group from 9,999 iterations of each null model to the model estimate for our observed data. In short, this null model tells us how strongly any randomly-chosen fraction (up to ~25%, depending on overlap between regional rarity and occurrence in the restoration dataset) of our species responds to habitat enhancement, providing scope to test whether rare species respond especially strongly to habitat enhancements. There are at least two reasons that we could fail to find a clear difference between the empirical model coefficients and those from the null models. The first of these is biological: the factors leading species to be regionally rare or common may not be strongly correlated to the factors that predispose species to respond strongly to habitat enhancement. The second of these is more related to classification accuracy: Even if *truly* regionally rare and common bee species respond differently to habitat enhancement than typical species do, we may have misclassified species as rare or, less likely, as common, obscuring the link between rarity/commonness and the propensity to benefit from habitat enhancement (Harrison *et al.* 2017). This null model analysis entailed generating nearly

264 120,000 mixed-effects models, and therefore, we did not complete model validation for the null
265 datasets.

266   We consider a group to clearly exhibit a stronger or weaker than expected response to
267 pollinator habitat restoration (e.g., abundance in pollinator planting vs. abundance in control)
268 when empirical model estimates are above 97.5% or below 2.5% of the null model results. This
269 analysis was conducted using R 3.4.1 (R Core Team 2017).

270 2.4.3 | Testing sensitivity of quartiles

271 To test whether our results were sensitive to the quantile threshold for what constitutes rarity (or
272 commonness), we re-ran our analyses and the null model for 10 additional thresholds, ranging
273 from the 20th to the 30th percentile. This analysis was conducted using R 3.4.1 (R Core Team
274 2017).

275

276 3 | Results

277   In the pollinator restoration dataset, we collected a total of 10,809 individual bees of 157
278 native species or species complexes. Of these, 195 specimens (37 species) were designated as
279 locally rare and 8,642 specimens (39 species) as locally common, based on the relative
280 abundance data set. Based on the site occurrence dataset, 65 specimens (14 species) were
281 designated as rare and 9,977 specimens of (77 species) were designated as common. For the
282 geographic range size metric, 316 specimens (34 species) were rare and 4081 specimens (40
283 species) were common.

284

285 3.1 | Do pollinator habitat restorations benefit rare and/ or common bees?

286   Species defined as rare based on local relative abundance had higher abundance and
287 richness? in the pollinator plantings compared to old-field controls by factors of nearly 3 and 2
288 (Table 1; Fig. 2A). Species categorized as rare based on low site occurrence frequency did not
289 exhibit a clear difference in the number species collected; however, the abundance of these rare
290 bees was reduced by 50% reduction in treatment versus controls (Table 1, Fig. 2B). Species
291 categorized as rare based on small geographic range size were not clearly more abundant or
292 species rich in pollinator plantings (Table 1, Fig. 2C). Regardless as to how we defined
293 commonness, we collected both more species and individuals from common bee species in
294 pollinator restorations compared to controls (Table 1, Fig. 3).

295

296 3.2 | Do pollinator habitat restorations lead to a disproportionate benefit for
297 either the rare or the common bees?

298   For almost all species rarity criteria and response types (richness or abundance), we
299 found no difference between the response of the species designated as rare, and results for a null
300 model that randomly chose an equally-sized group of species (Fig. 4). There was one exception:
301 habitat enhancements supported 54% lower abundance of bee species designated as rare based
302 on regional site occupancy (Fig. 4e, $p<0.01$). Similarly, for bee species designated as common

303   (for all commonness criteria) richness and abundance responses to habitat enhancement were not
304   different than expected for a random group of species (Fig. 5).

305

3.3 | Testing sensitivity of quartiles

307   The quantile sensitivity analysis mostly did not reveal qualitative differences in the
308   results (Table S3). However, as the quantile thresholds for rarity/commonness got larger (and
309   thus more inclusive), model coefficients tended towards the average for all species.

310

4 | Discussion

312

4.1 | Rare bee species exhibit atypical responses to habitat enhancements

314   Pollinator plantings, in general, increase the overall abundance and richness of wild,
315   native bees (reviewed in Tonietto & Larkin 2018; & Nicholson *et al.* 2020). We found that
316   common bee species and, depending on the criteria for designation, rare bee species had higher
317   abundance and species richness in wildflower plantings as compared to old-field controls. Our
318   results partially support other studies that found pollinator plantings benefit wild bees, including
319   rare and common species (Pywell *et al.* 2012; Scheper *et al.* 2015). For example, in the EU,
320   Scheper et al. (2015) found that all bee species, including those designated as rare by the IUCN,
321   increased in floral strips. Pollinator plantings likely benefit both rare and common species when
322   both groups overlap in the species of flowers they prefer, and these floral species are included in
323   the plantings (Macleod *et al.* 2020). As common species are among the most important crop
324   pollinators (Winfree *et al.* 2015), pollinator plantings that benefit them (even if they do not also
325   support rare species effectively) could have positive impacts on crop pollination (Blaauw &
326   Isaacs 2014; but see Nicholson *et al.* 2020; Lowe *et al.* 2021). However, depending on how bee
327   species were categorized as rare, and which response variable we tracked, pollinator plantings
328   failed to benefit all rare bees. In particular, bee species categorized as rare based on regional site
329   occurrence frequency had lower abundance in pollinator plantings as compared to old-field
330   controls. Overall, this indicates that, although pollinator plantings in general can have benefits
331   for biodiversity and ecosystem services, these two goals are not always met simultaneously
332   (Nicholson *et al.* 2020; Lowe *et al.* 2021), and not all species benefit from this conservation
333   action (Senapathi *et al.* 2015).

334   The effect of pollinator plantings on rare species depended on rarity criteria. Bee species
335   designated as rare based on local relative abundance were more abundant and species rich in
336   pollinator plantings. Even if a species has low local abundance, it may still benefit from
337   pollinator plantings if it occurs frequently across the landscape. This echoes findings elsewhere
338   that benefits of pollinator habitat enhancement are greatest in landscapes regions with
339   intermediate cover of natural and seminatural habitat, and are therefore likely to contain source
340   populations able to exploit additional resources (Grab *et al.* 2020; Griffin *et al.* 2021;
341   McCullough *et al.* 2021, but see Lane et al. 2020).

342   Conversely, species that do not commonly occur across a region may be unlikely to
343   benefit from pollinator plantings for two reasons. First, a number of rare bees are specialists and
344   collect pollen from a limited suite of plants. These species may be common when their host

345 species is present. Wildflower seed mixes used in pollinator plantings are only a small subset of
346 native plants within a region, inevitably fail to include important floral hosts. For example, at
347 one site, we caught seven *Trachusa dorsalis* (Megachilidae; Lepeletier 1841) in an old-field
348 control. This species had not been recorded in New Jersey or Pennsylvania (states where this
349 study occurred) for over 100 years (Ascher & Pickering 2019). *Trachusa dorsalis* specializes on
350 plants in the genus *Strophostyles* (Fabaceae), a genus not planted in the pollinator habitats we
351 sampled. Second, plantings are not targeted towards specific habitat types that may support rare
352 bees (Harrison *et al.* 2018). In our dataset, specialist bee species are likely to be categorized as
353 rare based on site occurrence frequency (assuming the resource on which they specialize is
354 heterogeneously available across the region), and potentially by approximate range size (if the
355 resource on which they specialize is also geographically restricted). We found that bee species
356 with low regional site occupancy showed much less of an abundance response to habitat
357 installation than we would expect for a random group of species, in line with our expectation that
358 pollinator plantings may not benefit uncommon specialist species with atypical resource
359 requirements. This finding echoes suggestions elsewhere (e.g. Macleod *et al.* 2020) that species-
360 specific management actions may be necessary if rare species conservation is an objective of
361 restoration efforts.

362

4.2 | Limitations

364 One limitation of our study is that the plantings we studied did not contain early-
365 blooming spring plants - a common problem in pollinator plantings (Wood et al. 2018).
366 Therefore, we were not able to assess whether pollinator plantings with spring plant species
367 might benefit spring bees. This may be particularly important in our region, as a number of early
368 spring species are known to be rare (Harrison *et al.* 2017).

369 A more general limitation is the inherent problems with categorizing species as rare in the
370 first place. Here we use a number of metrics and employ a quantile method to characterize
371 species rarity (Gaston 1994). This method has a number of important advantages: it is
372 straightforward, has a clear cut-off, and easily incorporates new data. One potential drawback is
373 the need to make an arbitrary decision about which quantile value to use for rarity designation
374 (Gaston 1994). Here we show that our results were robust to choice of quantile. However, there
375 are other important considerations when using this approach. For example, sampling biases (e.g.
376 in habitat type) in the reference datasets means the quantitative rankings might still not reflect
377 wider patterns of abundance and occurrence in nature.

378 We faced both misclassification risks and statistical limitations inherent to studying rare
379 species. Due to limited monitoring efforts for wild bees, it is not clear which species in our
380 region are truly rare, and classification errors are likely (Harrison *et al.* 2017). First, in defining
381 rarity, we chose to focus on records collected within a few hundred kilometers of our study sites,
382 and within a few weeks of the year of our sampling period, and thus consider species to be rare
383 even if they are common elsewhere or in other seasons. Second, due to limited data coverage, it
384 is likely that our rarity categorizations conflated rarity types: species with low abundance (for
385 example, specialized kleptoparasitic bees) could be mistakenly categorized as low occupancy
386 even if they are widespread in our study region, or species with strong habitat specialization or
387 low relative abundance over a large range could have erroneously small range-size estimates.
388 Furthermore, binary rarity classifications based on arbitrary thresholds (and for regional metrics,

389 arbitrary spatial scale) are likely to miss more biologically relevant definitions of rarity (Gaston
390 1994; Inger *et al.* 2015; Reed *et al.* 2020; Balbuena *et al.* 2021). Additionally, because they are
391 rare, any study of rare species is likely to face issues of statistical power and related concerns,
392 such as type-S and type-M errors (Gelman & Carlin 2014). In part, we were able to defend
393 against these issues with sensitivity analysis and our null model simulation. Yet, even as
394 qualitative patterns were robust to changing rarity thresholds, and null-model simulations, in
395 some cases, indicated that differing responses from rare species were not expected simply as a
396 result of classifying some species as rare and others as not rare, we urge caution in generalizing
397 our findings. Overall, our results demonstrate heterogeneity in response to habitat enhancements
398 at the species level, and many species are, of course, rare. Diverse taxonomic and functional
399 groups are unlikely to respond uniformly to conservation actions, and more targeted conservation
400 goals and strategies may be easier to assess and fine-tune.

401

4.3 | Conclusions and Management Implications

403 Currently, plant species used in pollinator plantings are chosen because of cost,
404 availability, or whether the seed mix attracts high pollinator species richness or abundance
405 (Harmon-Threatt & Hendrix 2015; M'Gonigle *et al.* 2017; Williams & Lonsdorf 2018).
406 However, it may be that current pollinator plantings primarily benefit species that already persist
407 within anthropogenic landscapes (Senapathi *et al.* 2015). If land managers aim to conserve rare
408 bees, they may need to incorporate plants not traditionally used in pollinator habitat plantings
409 (MacLeod *et al.* 2020).

410 Although they were the focus of our study, it is not necessarily the case that rare species
411 *should be* the target when designing and implementing pollinator plantings. While conservation
412 biology has focused on rarity, temporal trends (population decline, habitat loss, and range
413 contraction) may better predict extinction risk (Collen *et al.* 2016) – of course, these trends are
414 difficult to detect in rarely sampled species, such as many we have studied here. At the same
415 time, common species are also of critical conservation concern. In Europe, common bird species
416 are experiencing declines while less abundant species are increasing (Inger *et al.* 2015). Further,
417 common species may also have large effects on ecosystem function (Gaston 2011; Kleijn *et al.*
418 2015; Genung *et al.* 2018). More important than whether current practices benefit rare species,
419 ensuring overlap between conservation approaches and their likely impact on *particular* species
420 of concern may produce the best conservation outcomes, as disparate responses to habitat
421 alterations should be expected across diverse taxonomic groups.

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References

Albrecht, M., Kleijn, D., Williams, N.M., Tschumi, M., Blaauw, B.R., Bommarco, R., *et al.* (2020). The effectiveness of flower strips and hedgerows on pest control, pollination services and crop yield: a quantitative synthesis. *Ecol. Lett.*

Ascher, J.S. & Pickering, J. (2019). *Discover Life bee species guide and world checklist (Hymenoptera: Apoidea: Anthophila)*. *Discover Life bee species guide and world checklist (Hymenoptera: Apoidea: Anthophila).* Available at: http://www.discoverlife.org. Last accessed 2019.

Balbuena, J.A., Monlleó-Borrull, C., Llopis-Belenguer, C., Blasco-Costa, I., Sarabeev, V.L. & Morand, S. (2021). Fuzzy quantification of common and rare species in ecological communities (FuzzyQ). *Methods Ecol. Evol.*

Bartomeus, I., Ascher, J.S., Gibbs, J., Danforth, B.N., Wagner, D.L., Hedtke, S.M., *et al.* (2013). Historical changes in northeastern US bee pollinators related to shared ecological traits. *Proc. Natl. Acad. Sci. U. S. A.*, 110, 4656–4660.

Bates, D., Mächler, M., Bolker, B. & Walker, S. (2015). Fitting Linear Mixed-Effects Models Using lme4. *Journal of Statistical Software, Articles*, 67, 1–48.

Blaauw, B.R. & Isaacs, R. (2014). Flower plantings increase wild bee abundance and the pollination services provided to a pollination-dependent crop. *J. Appl. Ecol.*, 51, 890–898.

Blackburn, T.M. & Gaston, K.J. (1997). Who is rare? Artefacts and complexities of rarity determination. In: *The Biology of Rarity: Causes and consequences of rare—common differences* (eds. Kunin, W.E. & Gaston, K.J.). Springer Netherlands, Dordrecht, pp. 48–60.

Brudvig, L.A. (2017). Toward prediction in the restoration of biodiversity. *J. Appl. Ecol.*, 54, 1013–1017.

Carvell, C., Roy, D.B., Smart, S.M., Pywell, R.F., Preston, C.D. & Goulson, D. (2006). Declines in forage availability for bumblebees at a national scale. *Biol. Conserv.*, 132, 481–489.

Collen, B., Dulvy, N.K., Gaston, K.J., Gärdenfors, U., Keith, D.A., Punt, A.E., *et al.* (2016). Clarifying misconceptions of extinction risk assessment with the IUCN Red List. *Biol. Lett.*, 12.

Crone, E.E. & Williams, N.M. (2016). Bumble bee colony dynamics: quantifying the importance of land use and floral resources for colony growth and queen production. *Ecol. Lett.*, 19, 460–468.

Dushoff, J., Kain, M.P. & Bolker, B.M. (2019). I can see clearly now: Reinterpreting statistical significance. *Methods Ecol. Evol.*, 10, 756–759.

Fagan, W.F., Unmack, P.J., Burgess, C. & Minckley, W.L. (2002). Rarity, Fragmentation, and Extinction Risk in Desert Fishes. *Ecology*, 83, 3250–3256.

Garbach, K. & Long, R.F. (2017). Determinants of field edge habitat restoration on farms in California's Sacramento Valley. *J. Environ. Manage.*, 189, 134–141.

Gaston, K.J. (1994). *Rarity*. Population and Community Biology Series.

Gaston, K.J. (2011). Common Ecology. *Bioscience*, 61, 354–362.

Gelman, A. & Carlin, J. (2014). Beyond Power Calculations: Assessing Type S (Sign) and Type M (Magnitude) Errors. *Perspect. Psychol. Sci.*, 9, 641–651.

Genung, M.A., Fox, J., Williams, N.M., Kremen, C., Ascher, J., Gibbs, J., *et al.* (2017). The relative importance of pollinator abundance and species richness for the temporal variance of pollination services. *Ecology*, 98, 1807–1816.

Gibbs, J.P., Marquez, C. & Sterling, E.J. (2008). The role of endangered species reintroduction in ecosystem restoration: Tortoise-cactus interactions on espanola island, Galapagos.

478   *Restor. Ecol.*, 16, 88–93.

479 Grab, H., Poveda, K., Danforth, B. & Loeb, G. (2018). Landscape context shifts the balance of
480   costs and benefits from wildflower borders on multiple ecosystem services. *Proc. Biol. Sci.*,
481   285.

482 Griffin, S.R., Bruninga-Socolar, B. & Gibbs, J. (2021). Bee communities in restored prairies are
483   structured by landscape and management, not local floral resources. *Basic Appl. Ecol.*, 50,
484   144–154.

485 Hallett, L.M., Diver, S., Eitzel, M.V., Olson, J.J., Ramage, B.S., Sardinas, H., *et al.* (2013). Do
486   We Practice What We Preach? Goal Setting for Ecological Restoration. *Restor. Ecol.*, 21,
487   312–319.

488 Harmon-Threatt, A.N. & Hendrix, S.D. (2015). Prairie restorations and bees: The potential
489   ability of seed mixes to foster native bee communities. *Basic Appl. Ecol.*, 16, 64–72.

490 Harnik, P.G., Simpson, C. & Payne, J.L. (2012). Long-term differences in extinction risk among
491   the seven forms of rarity. *Proc. Biol. Sci.*, 279, 4969–4976.

492 Harrison, T., Gibbs, J. & Winfree, R. (2017). Anthropogenic landscapes support fewer rare bee
493   species. *Landsc. Ecol.*, 1–12.

494 Harrison, T., Gibbs, J. & Winfree, R. (2018). Forest bees are replaced in agricultural and urban
495   landscapes by native species with different phenologies and life-history traits. *Glob. Chang.*
496   *Biol.*, 24, 287–296.

497 Hartig, F. (2020). *DHARMa: Residual Diagnostics for Hierarchical (Multi-Level / Mixed)*
498   *Regression Models*.

499 Hobbs, R.J. (2007). Setting Effective and Realistic Restoration Goals: Key Directions for
500   Research. *Restor. Ecol.*, 15, 354–357.

501 Inger, R., Gregory, R., Duffy, J.P., Stott, I., Voříšek, P. & Gaston, K.J. (2015). Common
502   European birds are declining rapidly while less abundant species' numbers are rising. *Ecol.*
503   *Lett.*, 18, 28–36.

504 Kiessling, W. & Aberhan, M. (2007). Geographical distribution and extinction risk: lessons from
505   Triassic?Jurassic marine benthic organisms. *J. Biogeogr.*, 34, 1473–1489.

506 Kleijn, D., Winfree, R., Bartomeus, I., Carvalheiro, L.G., Henry, M., Isaacs, R., *et al.* (2015).
507   Delivery of crop pollination services is an insufficient argument for wild pollinator
508   conservation. *Nat. Commun.*, 6, 7414.

509 Lande, R. (1988). Genetics and demography in biological conservation. *Science*, 241, 1455–
510   1460.

511 Lane, I.G., Herron-Sweet, C.R., Portman, Z.M. & Cariveau, D.P. (2020). Floral resource
512   diversity drives bee community diversity in prairie restorations along an agricultural
513   landscape gradient. *J. Appl. Ecol.*, 57, 2010–2018.

514 Lowe, E.B., Groves, R. & Gratton, C. (2021). Impacts of field-edge flower plantings on
515   pollinator conservation and ecosystem service delivery – A meta-analysis. *Agric. Ecosyst.*
516   *Environ.*, 310, 107290.

517 Macleod, M., Reilly, J., Cariveau, D., Genung, M., Roswell, M., Gibbs, J., *et al.* (2020). How
518   much do rare and crop-pollinating bees overlap in identity and flower preferences? *Journal*
519   *of Applied Ecology*.

520 MacLeod, M., Reilly, J., Cariveau, D.P., Genung, M.A., Roswell, M., Gibbs, J., *et al.* (2020).
521   How much do rare and crop-pollinating bees overlap in identity and flower preferences? *J.*
522   *Appl. Ecol.*, 57, 413–423.

523 Mandelik, Y., Winfree, R., Neeson, T. & Kremen, C. (2012). Complementary habitat use by wild

524   bees in agro-natural landscapes. *Ecol. Appl.*, 22, 1535–1546.
525   Manne, L.L., Brooks, T.M. & Pimm, S.L. (1999). Relative risk of extinction of passerine birds
526   on continents and islands. *Nature*, 399, 258.
527   Manne, L.L. & Pimm, S.L. (2001). Beyond eight forms of rarity: which species are threatened
528   and which will be next? *Anim. Conserv.*, 4, 221–229.
529   McCullough, C.T., Angelella, G.M. & O'Rourke, M.E. (2021). Landscape Context Influences
530   the Bee Conservation Value of Wildflower Plantings. *Environ. Entomol.*
531   McGill, B.J., Etienne, R.S., Gray, J.S., Alonso, D., Anderson, M.J., Benecha, H.K., *et al.* (2007).
532   Species abundance distributions: moving beyond single prediction theories to integration
533   within an ecological framework. *Ecol. Lett.*, 10, 995–1015.
534   M'Gonigle, L.K., Williams, N.M., Lonsdorf, E. & Kremen, C. (2017). A tool for selecting plants
535   when restoring habitat for pollinators. *Cons Lett*, 10, 105–111.
536   Nicholson, C.C., Ward, K.L., Williams, N.M., Isaacs, R., Mason, K.S., Wilson, J.K., *et al.*
537   (2020). Mismatched outcomes for biodiversity and ecosystem services: testing the
538   responses of crop pollinators and wild bee biodiversity to habitat enhancement. *Ecol. Lett.*,
539   23, 326–335.
540   Ollerton, J., Erenler, H., Edwards, M. & Crockett, R. (2014). Extinctions of aculeate pollinators
541   in Britain and the role of large-scale agricultural changes. *Science*, 346, 1360–1362.
542   Payne, J.L. & Finnegan, S. (2007). The effect of geographic range on extinction risk during
543   background and mass extinction. *Proc. Natl. Acad. Sci. U. S. A.*, 104, 10506–10511.
544   Payne, J.L., Truebe, S., Nützel, A. & Chang, E.T. (2011). Local and global abundance associated
545   with extinction risk in late Paleozoic and early Mesozoic gastropods. *Paleobiology*, 37,
546   616–632.
547   Pywell, R.F., Heard, M.S., Bradbury, R.B., Hinsley, S., Nowakowski, M., Walker, K.J., *et al.*
548   (2012). Wildlife-friendly farming benefits rare birds, bees and plants. *Biol. Lett.*, 8, 772–
549   775.
550   Rabinowitz, D. (1981). Seven forms of rarity. In: *The Biological Aspects of Rare Plant
551   Conservation* (ed. Synge, H.). Wiley, pp. 205–217.
552   R Core Team. (2017). *R: A Language and Environment for Statistical Computing.* http://www.R-
553   project.org/.
554   R Core Team. (2021). *R: A Language and Environment for Statistical Computing.*
555   Reed, J.M., Nguyen, A., Owens, A.C.S. & Lewis, S.M. (2020). Linking the seven forms of rarity
556   to extinction threats and risk factors: an assessment of North American fireflies. *Biodivers.
557   Conserv.*, 29, 57–75.
558   Roulston, T.H. & Goodell, K. (2011). The role of resources and risks in regulating wild bee
559   populations. *Annu. Rev. Entomol.*, 56, 293–312.
560   Scheper, J., Bommarco, R., Holzschuh, A., Potts, S.G., Riedinger, V., Roberts, S.P.M., *et al.*
561   (2015). Local and landscape-level floral resources explain effects of wildflower strips on
562   wild bees across four European countries. *J. Appl. Ecol.*, 52, 1165–1175.
563   Senapathi, D., Biesmeijer, J.C., Breeze, T.D., Kleijn, D., Potts, S.G. & Carvalheiro, L.G. (2015).
564   Pollinator conservation - the difference between managing for pollination services and
565   preserving pollinator diversity. *Current Opinion in Insect Science*, 12, 93–101.
566   Sutter, L., Jeanneret, P., Bartual, A.M., Bocci, G. & Albrecht, M. (2017). Enhancing plant
567   diversity in agricultural landscapes promotes both rare bees and dominant crop-pollinating
568   bees through complementary increase in key floral resources. *J. Appl. Ecol.*, 54, 1856–1864.
569   Swarts, N.D. & Dixon, K.W. (2009). Terrestrial orchid conservation in the age of extinction.

570   *Ann. Bot.*, 104, 543–556.

571   Tonietto, R.K. & Larkin, D.J. (2018). Habitat restoration benefits wild bees: A meta-analysis. *J.*

572   *Appl. Ecol.*, 55, 582–590.

573   Williams, N.M. & Lonsdorf, E.V. (2018). Selecting cost-effective plant mixes to support

574   pollinators. *Biol. Conserv.*, 217, 195–202.

575   Williams, N.M., Regetz, J. & Kremen, C. (2012). Landscape-scale resources promote colony

576   growth but not reproductive performance of bumble bees. *Ecology*, 93, 1049–1058.

577   Williams, N.M., Ward, K.L., Pope, N., Isaacs, R., Wilson, J., May, E.A., *et al.* (2015). Native

578   wildflower plantings support wild bee abundance and diversity in agricultural landscapes

579   across the United States. *Ecol. Appl.*, 25, 2119–2131.

580   Winfree, R., W Fox, J., Williams, N.M., Reilly, J.R. & Cariveau, D.P. (2015). Abundance of

581   common species, not species richness, drives delivery of a real-world ecosystem service.

582   *Ecol. Lett.*, 18, 626–635.

583   Table 1. Model output of comparison between pollinator plantings and paired old-field controls

584   for all three rarity metrics. Proportion change is exponentiated model output of change in

585   pollinator planting treatments compared to old-field controls. Bold and asterisks represent significant

586   difference between pollinator planting and old filed controls at p < 0.05.

	Relative Abundance			Site Occurrence Frequency			Geographic R	
	Proportion Increase	Confidence Interval	p-value	Proportion Increase	Confidence Interval	p-value	Proportion Increase	Confidence Interva
Rare Species								
Richness	**1.95**	**1.14 – 3.33**	**0.014***	1.5	0.55 – 4.09	0.428	1.64	0.97 –
Abundance	**2.93**	**1.31 – 6.55**	**0.009***	**0.46**	**0.24 – 0.88**	**0.019***	1.6	0.70 –
Common Species								
Richness	**1.34**	**1.11 – 1.62**	**0.002***	**1.42**	**1.22 – 1.64**	**<0.001***	1.43	1.17 –
Abundance	**2.78**	**1.77 – 4.39**	**<0.001***	**2.67**	**1.72 – 4.14**	**<0.001***	3	1.72 –

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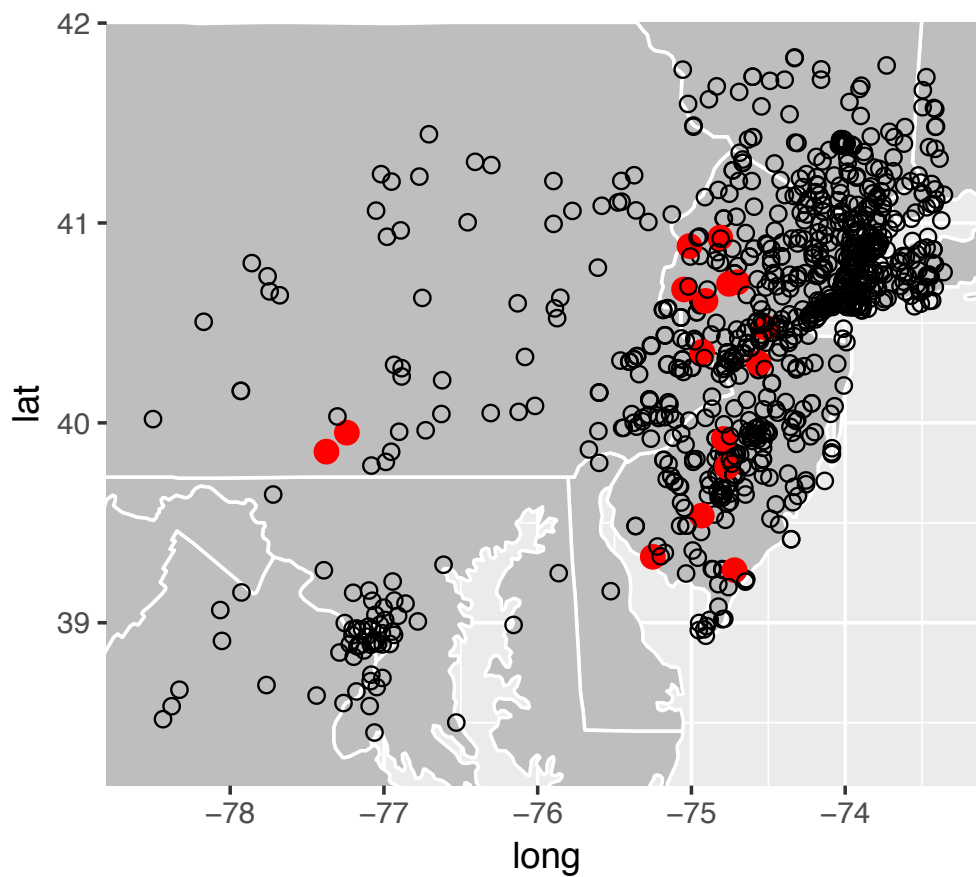
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591   Figure 1. Map of wildflower plantings (solid red circles) and locations of specimen records from
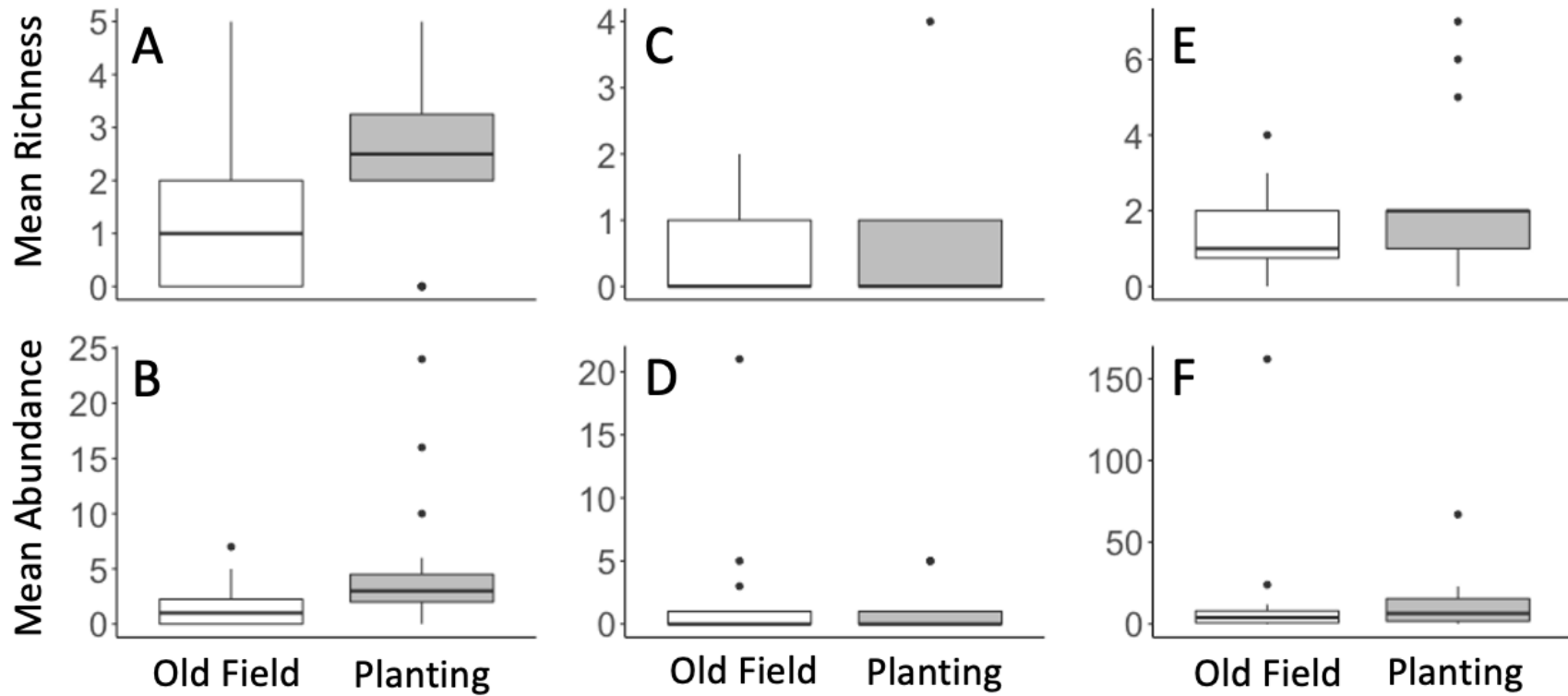592   independent regional dataset (open black circles).
593
594

Figure 2. Rare species richness and abundance in old fields (white) and pollinator habitat (grey). Rarity metrics for A-B) Relative abundance, C-D) site occurrence frequency, and E-F) geographic range size.
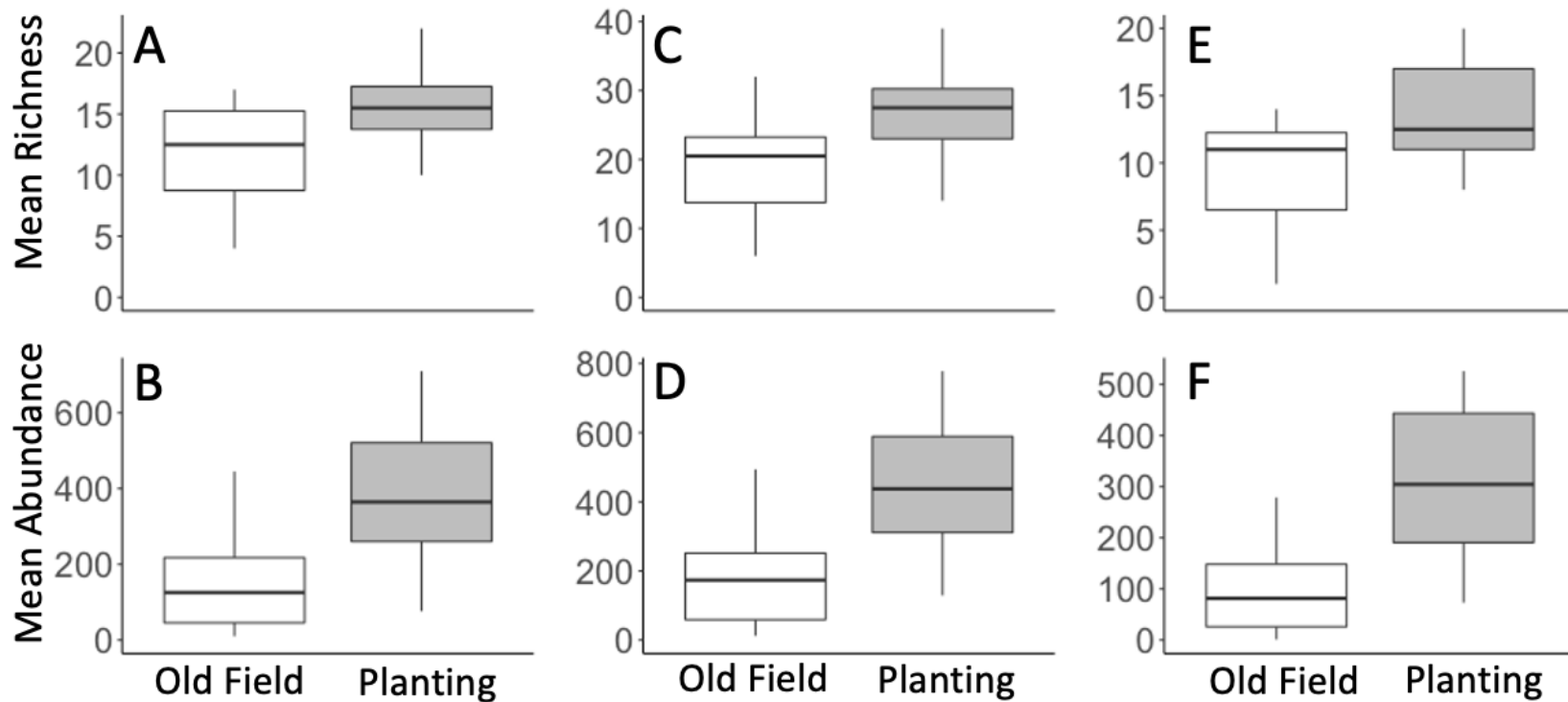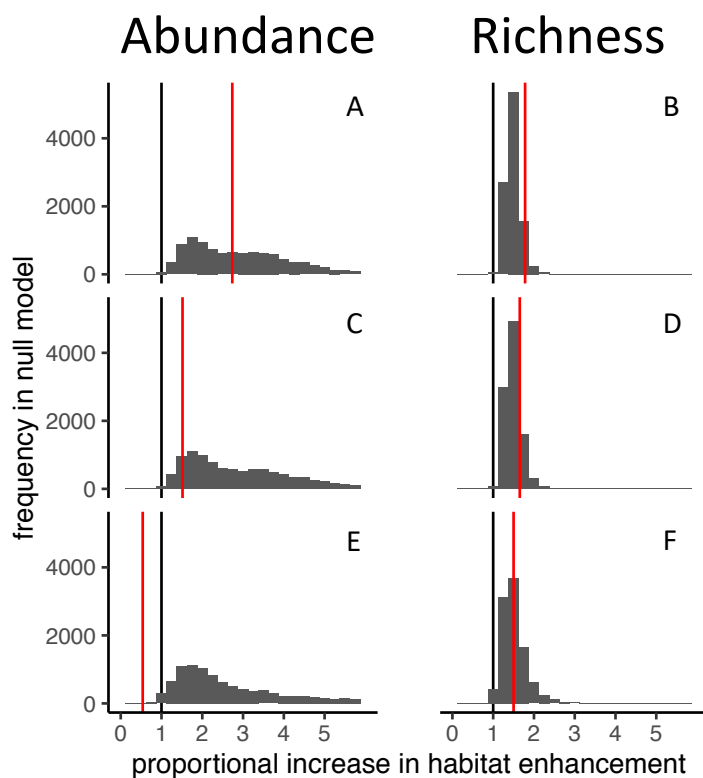
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Figure 3. Common species richness and abundance in old fields (white) and pollinator habitat (grey). Rarity metrics for A-B) Relative abundance, C-D) site occurrence frequency, and E-F) geographic range size.
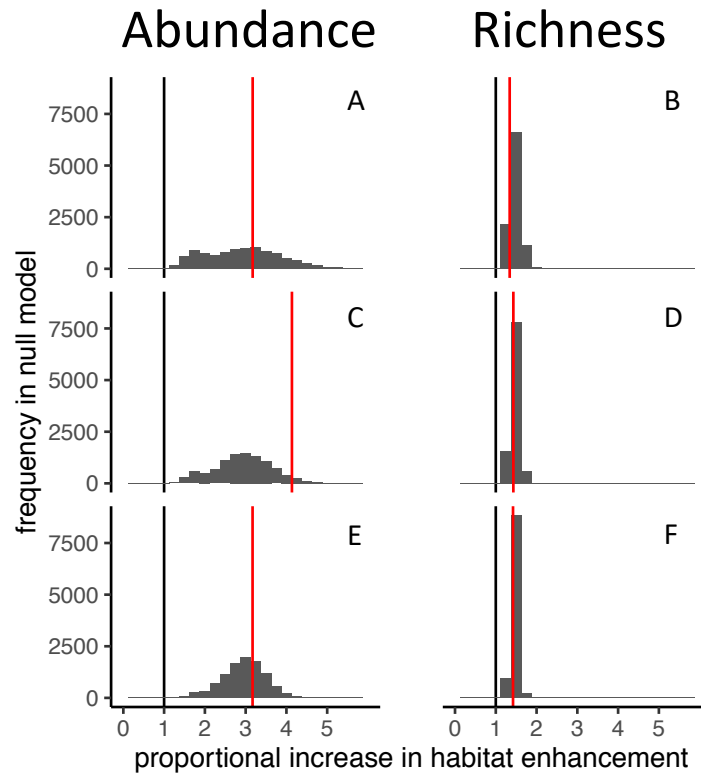
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612   Figure 4. Comparison of outputs for model simulations and empirical models for rare
613   species. x-axis values represent proportional increase in abundance or richness in habitat
614   enhancements relative to old field controls. Histograms represent the frequency of each
615   proportional change in richness or abundance in the null model, which designates randomly
616   selected species as "rare." Black lines represent no difference in response variable between
617   treatment and control (x=1). Red lines represent model output for the real data, for which
618   species were defined as rare based on regional relative abundance (A,B), regional site
619   occurrence frequency (C,D), or geographic range size (E, F).
620

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623 Figure 5. Comparison of outputs for model simulations and empirical models for common
624 species. x-axis values represent proportional increase in abundance or richness in habitat
625 enhancements relative to old field controls. Histograms represent the frequency of each
626 proportional change in richness or abundance in the null model, which designates randomly
627 selected species as "common." Black lines represent no difference in response variable
628 between treatment and control (x=1). Red lines represent model output for the real data, for
629 which species were defined as common based on regional relative abundance (A,B),
630 regional site occurrence frequency (C,D), or geographic range size (E, F).

631 **Table S1. Supplementary Information on the datasets used to classify rarity**

632 **Dataset Name**: dryad_amnh
633 Use in manuscript: site occurrence frequency
634 Number bee individual specimens used in this study: 7633
635 Number bee species used in this study: 331
636 Number of sites: 751
637 Published in: (Bartomeus *et al.* 2011, 2013a, b)
638
639 **Dataset Name**: bef_scale_spec
640 Use in manuscript: site occurrence frequency, local relative abundance
641 Number bee individual specimens used in this study: 7359
642 Number bee species used in this study: 90
643 Number of sites: 25
644 Unpublished
645
646 **Dataset Name**: cape_may
647 Use in manuscript: site occurrence frequency, local relative abundance
648 Number bee individual specimens used in this study: 8217
649 Number bee species used in this study: 85
650 Number of sites: 1
651 Published in: (MacLeod *et al.* 2016, 2020; Genung *et al.* 2017)
652
653 **Dataset Name**: forest_spec
654 Use in manuscript: site occurrence frequency, local relative abundance
655 Number bee individual specimens used in this study: 543
656 Number bee species used in this study: 59
657 Number of sites: 33
658 Published in: (Smith *et al.* In Press.; Volenec & Smith 2021)
659
660 **Dataset Name**: male_bee_project
661 Use in manuscript: site occurrence frequency, local relative abundance
662 Number bee individual specimens used in this study: 15110
663 Number bee species used in this study: 145
664 Number of sites: 8
665 Published in: (Roswell *et al.* 2019a, b)
666
667 **Dataset Name**: njpa_ha
668 Use in manuscript: site occurrence frequency, local relative abundance
669 Number bee individual specimens used in this study: 4822
670 Number bee species used in this study: 72
671 Number of sites: 54
672 Unpublished

673

674 **Dataset Name**: nsf0607_spec
675 Use in manuscript: site occurrence frequency, local relative abundance
676 Number bee individual specimens used in this study: 683
677 Number bee species used in this study: 70
678 Number of sites: 18
679 Published in: (Winfree *et al.* 2014)
680

681 **Dataset Name**: pinelands_2003
682 Use in manuscript: site occurrence frequency, local relative abundance
683 Number bee individual specimens used in this study: 1967
684 Number bee species used in this study: 124
685 Number of sites: 44
686 Published in: (Winfree *et al.* 2007)
687

688 **Dataset Name**: bh_spec
689 Use in manuscript: site occurrence frequency, local relative abundance
690 Number bee individual specimens used in this study: 5641
691 Number bee species used in this study: 162
692 Number of sites: 27
693 Published in: (Harrison *et al.* 2017, 2018b, a)
694

695 **Dataset Name**: swg
696 Use in manuscript: site occurrence frequency, local relative abundance
697 Number bee individual specimens used in this study: 7943
698 Number bee species used in this study: 192
699 Number of sites: 51
700 Unpublished
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708 Bartomeus, I., Ascher, J.S., Gibbs, J., Danforth, B.N., Wagner, D.L., Hedtke, S.M., *et al.*
709   (2013a). Historical changes in northeastern US bee pollinators related to shared
710   ecological traits. *Proc. Natl. Acad. Sci. U. S. A.*, 110, 4656–4660.
711 Bartomeus, I., Ascher, J.S., Gibbs, J., Danforth, B.N., Wagner, D.L., Hedtke, S.M., *et al.*
712   (2013b). Historical changes in northeastern US bee pollinators related to shared
713   ecological traits. Dryad Dataset: 10.5061/dryad.0nj49.
714 Bartomeus, I., Ascher, J.S., Wagner, D., Danforth, B.N., Colla, S., Kornbluth, S., *et al.* (2011).
715   Climate-associated phenological advances in bee pollinators and bee-pollinated plants.
716   *Proc. Natl. Acad. Sci. U. S. A.*, 108, 20645–20649.

717   Genung, M.A., Fox, J., Williams, N.M., Kremen, C., Ascher, J., Gibbs, J., *et al.* (2017). The
718   relative importance of pollinator abundance and species richness for the temporal
719   variance of pollination services. *Ecology*, 98, 1807–1816.
720   Harrison, T., Gibbs, J. & Winfree, R. (2017). Anthropogenic landscapes support fewer rare bee
721   species. *Landsc. Ecol.*, 1–12.
722   Harrison, T., Gibbs, J. & Winfree, R. (2018a). Forest bees are replaced in agricultural and urban
723   landscapes by native species with different phenologies and life-history traits. *Glob.*
724   *Chang. Biol.*, 24, 287–296.
725   Harrison, T., Gibbs, J. & Winfree, R. (2018b). Phylogenetic homogenization of bee communities
726   across ecoregions. *Glob. Ecol. Biogeogr.*, 27, 1457–1466.
727   MacLeod, M., Genung, M.A., Ascher, J.S. & Winfree, R. (2016). Measuring partner choice in
728   plant-pollinator networks: using null models to separate rewiring and fidelity from
729   chance. *Ecology*, 97, 2925–2931.
730   MacLeod, M., Reilly, J., Cariveau, D.P., Genung, M.A., Roswell, M., Gibbs, J., *et al.* (2020).
731   How much do rare and crop-pollinating bees overlap in identity and flower preferences?
732   *J. Appl. Ecol.*, 57, 413–423.
733   Roswell, M., Dushoff, J. & Winfree, R. (2019a). Male and female bees show large differences in
734   floral preference. *PLoS One*, 14, e0214909.
735   Roswell, M., Dushoff, J. & Winfree, R. (2019b). Male and female bees show large differences in
736   floral preference. Dryad Dataset. https://doi.org/10.5061/dryad.c3rr6q1. *Male and female*
737   *bees show large differences in floral preference*.
738   Smith, C., Harrison, T., Gardner, J. & Winfree, R. (In Press). Forest-associated bee species
739   persist amid forest loss and regrowth in eastern North America. *Biol. Conserv.*
740   Volenec, Z.M. & Smith, C.M. (2021). Not all matrix habitat is created equal for rare bee species
741   in forest habitat. *Ecol. Entomol.*
742   Winfree, R., Griswold, T. & Kremen, C. (2007). Effect of human disturbance on bee
743   communities in a forested ecosystem. *Conserv. Biol.*, 21, 213–223.
744   Winfree, R., Williams, N.M., Dushoff, J. & Kremen, C. (2014). Species abundance, not diet
745   breadth, drives the persistence of the most linked pollinators as plant-pollinator networks
746   disassemble. *Am. Nat.*, 183, 600–611.
747