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1 ~~With or without frugivores? Flying foxes sustain woody plant~~
2 ~~regeneration~~ **Differing severity of frugivore loss contrasts the fate of native**
3 **forests on the land of the Dodo (Mascarene archipelago)**

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1 Differing severity of frugivore loss contrasts the fate of native forests on the 2 land of the Dodo (*Mascarene archipelago*)

3 **ABSTRACT**

4 Frugivore loss has the potential to alter the regeneration of tropical hyperdiverse forests. However, no
5 study has investigated tree diversity maintenance within old-growth protected forests on tropical
6 oceanic islands where conservation stakes are considerable. In the Mascarenes, the largest frugivores
7 including the Dodo went extinct after human colonization in the 17th century. Mauritius, today devoid of
8 almost all its native habitats, retains an important population of flying foxes, while Réunion, largely
9 covered by native habitats, has lost all but one of its native frugivores, a bulbul. Using census data of
10 130 woody species from 3-ha permanent plots from Mauritius (MAU-BF) and Réunion (REU-ML), we
11 show that most fleshy-fruited species regenerate poorly at REU-ML, in contrast to MAU-BF, although
12 large-seeded fleshy-fruited species regenerate poorly at both sites. The difference in recruitment is all
13 the more striking because local woody plant diversity at MAU-BF is double that at REU-ML. Changes
14 observed in the understory will probably lead to a strongly impoverished canopy at REU-ML, and to
15 canopy shift composition at MAU-BF. Hence, the protected but highly fragmented forests on Mauritius
16 are likely more resilient than those on Réunion usually considered best preserved within the ecoregion.
17 Although processes other than seed dispersal might also be at work, flying foxes probably ensure
18 effective dispersal of numerous native fleshy-fruited plants on Mauritius, contrary to only bulbuls on
19 Réunion. To efficiently protect tropical rainforests on islands and continents alike, there is an urgent
20 need to protect extant frugivores populations and reinstate seed dispersal function wherever necessary.

21 **KEYWORDS**

22 Frugivore extinction; island biology; Janzen-Connell effect; plant diversity loss; protected tropical forest;
23 seed dispersal disruption

24 **INTRODUCTION**

25 Vertebrates play an important role as mutualistic pollinators or seed dispersers in most tropical forests
26 worldwide (Dirzo et al., 2014; Turner, 2001). Recently, it has become clear that their loss has the
27 potential to deeply alter tropical forest composition (Albert et al., 2020b; Dirzo et al., 2014; Effiom et al.,
28 2013; Terborgh et al., 2008; Trolliet et al., 2019; Wandrag et al., 2017). Most conservation strategies
29 have for decades involved integrating forest habitats into protected areas. However, preserving forests
30 in which mutualistic vertebrates are absent or at low density might be ineffective when their functioning
31 highly depends on plant-animals interactions (Laurance et al., 2012).

32 The various life-history strategies of trees complicate the assessment of the impact of frugivore
33 extirpation, but studies at community level indicate an alteration of the recruitment for numerous fleshy-
34 fruited plants in defaunated protected forests of the Neotropics (Terborgh et al., 2008), Central Africa
35 (Effiom et al., 2013; Trolliet et al., 2019) and Borneo (Harrison et al., 2013). Changes in recruitment
36 were detected just a few decades after frugivore decline or loss, being mainly the consequence of the
37 negative distance- and/or density-dependent effects on germination or seedling survival close to
38 maternal trees (Comita et al., 2014; Connell, 1971; Janzen, 1970). Other works have also shown how
39 fruit consumption by frugivores may enhance the recruitment of fleshy-fruited plants by removing
40 potential physical/chemical dormancy (Soltani et al., 2018; Traveset et al., 2008) or more simply, by
41 cleaning the seed of flesh that could attract pathogens and seed predators causing propagule mortality
42 (Levi and Peres, 2013). Hence, frugivores play a central role in closing the loop of dispersal in habitats
43 largely dominated by fleshy-fruited plants (Wang and Smith, 2002).

44 In theory, the decline and extirpation of frugivores threaten tropical forests not only on
45 continents, where they can be hyperdiverse (Terborgh et al., 2008), but also on oceanic islands

46 (Hansen et al., 2008). Despite their isolation and small areas, oceanic islands can indeed host
47 substantial richness in fleshy-fruited species, e.g. one forest patch on Mauritius harbours more than 60
48 tree species per hectare of which 89% are fleshy-fruited (Florens et al., 2017a, 2012). This relative
49 diversity is comparable to some African continental tropical rainforests (Turner, 2001). Hence, one may
50 expect a higher negative impact of frugivore loss on diversity maintenance on tropical oceanic islands
51 since numerous frugivores went extinct after human colonization (Hansen and Galetti, 2009; Heinen et
52 al., 2017), and having no options of alternative seed dispersal systems (Guimarães et al., 2008).

53 Fleshy-fruited plant species vary in their vulnerability following the extinction of frugivores
54 (Guimarães et al., 2008). Plants with large fruits and/or large seeds are particularly at risk because
55 large vertebrates often constitute the first guild to be extirpated (Dirzo et al., 2014; Hansen and Galetti,
56 2009; Heinen et al., 2017). Several studies have shown that insular large-seeded tree species
57 experience recruitment failure at plant population level due to disperser loss (Carpenter et al., 2018;
58 Hansen et al., 2008; Wotton and Kelly, 2011). Although large-seeded species are reputedly well
59 adapted to the strong competition encountered in tropical understory (Leishman et al., 2000), their
60 seedlings/saplings can be strongly depleted at community level in defaunated sites (Effiom et al., 2013;
61 Harrison et al., 2013; Terborgh et al., 2008; Trollet et al., 2019). To our knowledge however, the
62 relationship between recruitment patterns inferred from size-distributions and defaunation has never
63 been carried out before on islands, where one may expect a strong negative impact of large-bodied
64 frugivore extinction on large-seeded plant recruitment.

65 This study aims to assess the impact of frugivore extinction on forest regeneration by comparing
66 diameter cohorts of native woody species within two islands sharing a close biogeographic history, but
67 with different trajectories since human colonization (Fig.1). Mauritius and Réunion had lowland
68 species-rich forests dominated by fleshy-fruited species (Albert et al., 2018; Florens et al., 2017a;
69 Strasberg, 1996). On Réunion, the extirpation of large frugivores from Réunion dates back two
70 centuries (Fig.1). Consequently, primary succession dynamics have been severely altered after
71 frugivore extinctions and great uncertainty remains about the capacity of established forests to maintain
72 their diversity in the near future (Albert et al., 2020b). On Mauritius by contrast, *Pteropus niger* still
73 exists in large populations and is a keystone species which feeds fruits of at least 53% of woody plants
74 (Florens et al., 2017a; Fig.1). With a mean body mass of 450 g, this flying-fox is about nine times larger
75 than the bulbul *Hypsipetes borbonicus*, the largest extant frugivorous bird on Réunion (Cheke and
76 Hume, 2008). This critical difference in the defaunation history between the islands provides a relevant
77 opportunity to use these two main islands of the Mascarenes as pseudo-replicates, and to isolate the
78 effects of seed dispersal from other factors on plant recruitment. Although this biogeographical setting
79 is relevant to address the effects of defaunation on forest regeneration, we note that Mauritius cannot
80 be considered as a strict control in terms of frugivory interaction. Indeed, Mauritius also lost its largest
81 frugivore species that were able to swallow large fruits (Fig.1) and their extirpation has probably not
82 been fully compensated for by the surviving fauna (Howe, 2016; Meehan et al., 2002; Trollet et al.,
83 2019).

84 Using census data from permanent plots at Brise Fer on Mauritius (MAU-BF) and Mare Longue
85 on Réunion (REU-ML), we address three complementary hypotheses in response to differential
86 alteration of seed dispersal networks. First, we assume that plant diversity is better maintained at MAU-
87 BF than at REU-ML. In particular, at community level, we expect that the diversity among juveniles
88 should be similar or greater than that of adults at MAU-BF, whereas it should be lower at REU-ML.
89 Second, within species, we assume a higher number of juveniles relative to adults at MAU-BF than at
90 REU-ML. Indeed, few juveniles in proportion at a given time may indicate a declining population (Condit
91 et al., 1998). Third, we hypothesize seed size to have a strong influence on plant diversity and juveniles
92 ratio. The extinction of all Mascarene frugivores able to swallow fruit >13 mm in diameter probably led
93 to recruitment failure of large-seeded fleshy-fruited plants on both islands, although the extant flying
94 foxes might have, to some extent, carried out the function of these extinct large frugivores on Mauritius.

95 However, some plants, especially small-seeded species, might also have difficulties in regenerating in
96 the understory independently of their ability to be dispersed, because of adverse establishment
97 conditions, including competition and shade (Howe and Smallwood, 1982; Leishman et al., 2000). This
98 could be particularly true at REU-ML where the canopy is higher and the understory darker than at
99 MAU-BF.

100 **METHODS**

101 **Study sites**

102 Réunion (2512 km², 2-3 million years) and Mauritius (1865 km², 7.8 million years), the two largest
103 islands of the Mascarene archipelago in the tropical Indian Ocean, are respectively 750 and 900 km
104 east of Madagascar (Fig.1). They are part of the biodiversity hotspot of South-West Indian Ocean
105 islands (Mittermeier et al., 2004). The Mascarene islands share most of their animal and plant lineages,
106 with a great number of endemic species given their relatively small terrestrial area (Kier et al., 2009).
107 Mauritius and Réunion host 691 native flowering plant species (273 single island endemic and 150
108 Mascarene endemic) and 550 species (165 single-island endemic and 140 Mascarene endemic),
109 respectively (Baider et al., 2010). Extinct and threatened species make 10.9% and 81.7% on Mauritius,
110 while 1.2% and 50.9% on Réunion (Baider et al., 2010). Lower rates of plant extinction and threat on
111 Réunion is attributed to its higher native cover (30%) than on Mauritius (4.4%), although lowland
112 habitats have been almost completely transformed on Réunion (Florens et al., 2012; Strasberg et al.,
113 2005). In that context, permanent plots at Brise Fer on Mauritius (MAU-BF) and Mare Longue on
114 Réunion (REU-ML) are used for conservation purposes to monitor the remnants of evergreen wet
115 tropical rainforest in the Mascarenes (Cadet, 1977; Florens et al., 2012; Safford, 1997; Strasberg et al.,
116 2005).

117 *Mare Longue (Réunion)*

118 The three 1-ha plots at REU-ML (centred around 21°21'13" S, 55°44'40" E) were in reserve (68
119 hectares), now included in the National park of Réunion (Fig.1). This site is at the lower reaches of the
120 last continuous corridor of vegetation in the Mascarenes running from 100 m asl to the volcano caldera
121 at 2400 m. This forest corridor is surrounded by secondary vegetation and sugar cane fields at lower
122 elevations. The studied plots are situated between 130 and 340 m asl and receive ca 4000 mm of mean
123 annual rainfall (Réchou et al., 2019; Tab.1). The forest at REU-ML is classified as a lowland tropical
124 rainforest with no deciduous species (Cadet, 1977) and is little invaded by alien plants . It is set on
125 pahoehoe basaltic lava flow estimated at ca 430 years old (Albert et al., 2020a) and not yet significantly
126 altered. Emergent trees can reach 25 m high and canopy is about 15-20 m, with an average of ca 900
127 stems >10 cm dbh per hectare and basal area over 70 m².ha⁻¹ (Tab.1).

128 *Brise Fer (Mauritius)*

129 The three 1-ha plots at MAU-BF (centred around 20°22'20" S, 57°26'10" E) are near the western edge
130 of Mauritius central plateau, in the Black River Gorges National Park (Fig.1). This National Park has the
131 largest remnants of native forests on Mauritius (Florens, 2013). The studied plots are located at 560-
132 600 m asl and receive ca 2400 mm of mean annual rainfall (Tab.1). The forest at MAU-BF is classified
133 as a tropical rainforest (Lorence and Sussman, 1986). Trees can reach 18 m high and canopy is about
134 12-15 m. MAU-BF has 1000-1600 stems >10 cm dbh per hectare but lower basal area than REU-ML
135 (Tab.1). The area has been weeded of alien plant species since 1996 (Baider and Florens, 2011) .

136 **Sampling**

137 Monitoring of permanent plots began in the 1990s (at REU-ML), but we used the data sampled between
138 2010 and 2014 to avoid temporal bias. We used saplings reaching 1 cm diameter at breast high (dbh,

139 measured at 1.3 m), as this was the threshold for measurements at REU-ML. We calculated an
140 equivalent dbh for plants that had multiple stems.

141 *Growth forms*

142 We categorized the growth form of each woody species as either tree, shrub or climber according to the
143 description in the *Flore des Mascareignes* (Bossier et al., 1976–ongoing) or newer publication, and
144 expert knowledge of the local floras. Shrubs could reach up to 15 cm dbh, while trees reached a
145 maximum of 94 cm at MAU-BF and 148.8 cm dbh at REU-ML.

146 *Diameter cohorts*

147 We excluded climbers and species without secondary growth (e.g. Monocots, fern trees) from the
148 analyses and defined “young”, “intermediate” and “adult” cohorts as stems with dbh of [1; 3[cm, [3; 10]
149 cm and >10 cm, respectively for trees, and [1; 1.5[cm, [1.5; 3] cm and >3 cm, respectively for shrubs
150 (Fig.2). For each site, we extracted the number of young, intermediate and adult cohorts for each
151 species.

152 *Dispersal traits*

153 We categorized the fruit type of each woody species as either fleshy (FF) or dry (DF), using the *Flore*
154 *des Mascareignes* (Bossier et al., 1976-ongoing), and expert knowledge of the local floras when
155 information was missing or incomplete. Fruits were classified as fleshy if they had noticeable fleshy
156 pericarps or fleshy appendages when mature. In the Mascarenes, most DF plants rely on abiotic agents
157 for their dispersal.

158 We used seed length as a proxy for seed size as this trait was known for all species, based on
159 the *Flore des Mascareignes* (Bossier et al., 1976-onwards) and authors' field measurements. At MAU-
160 BF and REU-ML, seed length of woody species ranged from 0.06 cm (*Weinmannia tinctoria*,
161 Cunoniaceae) to 5.3 cm (*Elaeocarpus integrifolius*, Elaeocarpaceae), which represented an order of
162 magnitude of 10^6 in seed mass. We assigned seed length to three size classes (Appendix Fig.A1):
163 small-seeded plants [0.06; 0.6] cm, medium-seeded plants [0.6; 1.5] cm and large-seeded plants [1.5;
164 5.3] cm (following Corlett and Primack, 2006; Costa et al., 2012). Large seeds cannot be swallowed by
165 the extant frugivores in the Mascarenes, but can be dispersed by flying foxes over short distances at
166 MAU-BF (Florens et al., 2017a). Because there were no large-seeded and few medium-seeded DF
167 species, we combined both classes to create a “dispersal trait” factor with four classes: (1) all DF
168 species; and FF species that were (2) small-seeded, (3) medium-seeded, and (4) large-seeded.

169 **Analyses**

170 *Dominance-diversity across diameter cohorts*

171 To analyze changes in woody species dominance-diversity across diameter cohorts (i.e. from adults to
172 young), we used a rank-abundance distribution approach. Rank-abundance distributions display log-
173 transformed species abundance ranked in declining order of species abundance. This approach,
174 recommended by Matthews and Whittaker (2015), allows to describe the full distribution of
175 commonness and rarity for the whole woody community and according dispersal traits at each site.

176 *Diversity indices across diameter cohorts*

177 We first used the Shannon diversity index ($I_{shannon}$) to synthetically characterize changes in diversity
178 highlighted with rank-log abundance plots. We then assessed changes in phylogenetic diversity using
179 the function *pla* (R package 'adiv') that generalizes PD index of Faith (1992) to account for species
180 abundance (Pavoine, 2020). In this respect, we constructed phylogenetic trees for woody plant
181 communities based on the phylogeny from Zanne et al. (2013) (R package 'brranching').

182 Phylogenetic decoupling can show whether changes in diversity affect particular lineages.
183 Phylogenetic decoupling was calculated as $(p_{a=1, adult} / p_{a=1, young}) / (I_{shannon, adult} / I_{shannon, young})$ and $(p_{a=1, adult} / p_{a=1, int.}) / (I_{shannon, adult} / I_{shannon, int.})$ for “adult vs young cohorts” and “adult vs intermediate cohorts”
184 respectively. We assume values >1.1 and <0.9 would reveal stronger and weaker changes in
185 phylogenetic diversity than in Shannon diversity, respectively.
186

187 *Demographic status assessment*

188 The demographic status of a tree population can be assessed via the ratio of juveniles to adults (Condit
189 et al., 1998), hereafter called recruitment index. We therefore calculated the recruitment index of each
190 woody species: $n_{young} / (n_{young} + n_{adult})$ for youngs and $n_{int} / (n_{int} + n_{int})$ for intermediate cohorts, including
191 rare species, e.g. a single canopy tree without any conspecific individual in the understory. Finally, we
192 estimated, at each site, the distribution of the recruitment index within classes of dispersal traits and
193 growth forms (shrubs vs. trees). Analyses were done in R ver. 4.0.2 (R Core Team).

194 **RESULTS**

195 Total woody species richness was higher at Brise Fer (MAU-BF, 100 species: 65 trees, 35 shrubs) than
196 at Mare Longue (REU-ML, 51 species: 42 trees, 9 shrubs). Woody plants were overwhelmingly fleshy-
197 fruited (FF) (91% at MAU-BF; 88% at REU-ML). MAU-BF and REU-ML shared only 20 of the 130
198 species studied in total, but shared 35 genera (51% of those at MAU-BF and 81% at REU-ML; Tab.A2).
199 Eight other genera present at REU-ML, but absent at MAU-BF do exist on Mauritius. Among the 31
200 genera present at MAU-BF but absent at REU-ML, 19 were present on Réunion (often in subhumid
201 habitats, e.g. *Pleurostyliia*, *Pyrostria*, *Securinega*, *Scutia*) and 12 only occurred on Mauritius (Tab.A2).
202 MAU-BF showed 1.47 species per genus on average against 1.16 species per genus at REU-ML, as a
203 result of the existence of various radiated plant species at MAU-BF.

204 **Dominance-diversity across diameter cohorts**

205 Across diameter cohorts (from adults, to intermediate cohorts and youngs), woody species richness
206 strongly decreased from 47, to 40 and 35 species at REU-ML, whereas it slightly decreased at MAU-BF
207 from 87, to 84 and 80 species (Fig.3). We detected a strong increase in the steepness of the
208 dominance-diversity curves of woody species overall at REU-ML. Some species excessively dominated
209 young cohorts, and rare species were more numerous among intermediate and young cohorts despite
210 depauperate richness. On the contrary, we detected no increase in steepness in the dominance-
211 diversity curves at MAU-BF, but a tailing-off of rare species (Fig.3).

212 Considering dispersal traits (DF plants, FF small-, medium- and large-seeded plants), all groups
213 were negatively impacted at REU-ML, seen by a noticeable decline in species richness across diameter
214 cohorts (Fig.3). The increase in steepness curves of FF plants was related to one or several species
215 becoming more dominant, e.g. the shrub *Chassalia corallioides* (Rubiaceae) and the tree
216 *Labourdonnaisia calophylloides* (Sapotaceae) among medium-seeded and large-seeded plants,
217 respectively. Actually, most ranks among young cohorts at REU-ML showed a strong deficit of
218 abundance in comparison to adults, whatever dispersal traits (Fig.3).

219 At MAU-BF, FF small- and medium-seeded plants showed stability and a slight increase of
220 species richness across diameter cohorts, respectively. Both groups had higher juvenile abundance on
221 most ranks (Fig.3). Conversely, the strong increase in the steepness of dominance-diversity curve of FF
222 large-seeded plants was accompanied by an important decline in species richness (-8 species; Fig.3).
223 Large-seeded species were diverse in the canopy of MAU-BF with five species exceeding 100 stems
224 (surprisingly, only one species at REU-ML), but their stem proportion strongly decreased across
225 diameter cohorts from 0.37 to 0.21. Hence, the tailing-off of the overall dominance-diversity curve at
226 MAU-BF was mainly due to diversity loss of large-seeded plants.

227 Dry-fruited plants showed low species richness at both REU-ML and MAU-BF, but they were

228 nevertheless fairly well represented in the canopies of both forests (Fig.3). While dominance-diversity
229 curves were relatively similar across diameter cohorts at MAU-BF, dry-fruited plants were rare among
230 juveniles at REU-ML. This shows that these plants could not be involved in any replacement in the
231 understory at both sites and, therefore, were not studied further below.

232 **Diversity indices across diameter cohorts of fleshy-fruited plants**

233 Similarly to dominance-diversity curves, Shannon indices of small and medium-seeded plants strongly
234 decreased across diameter cohorts at REU-ML, while they increased at MAU-BF (Fig.4). The Shannon
235 index of large-seeded plants, which was already low for adults at REU-ML, declined further, and
236 steadily decreased at MAU-BF. Phylogenetic diversity showed broadly similar trends across diameter
237 cohorts for most dispersal traits at both sites. However, phylogenetic decoupling, i.e. values > 1.1 and $<$
238 0.9 (Fig.4), was observed for medium-seeded plants at REU-ML and large-seeded plants at MAU-BF.
239 At MAU-BF, the lower decrease in phylogenetic diversity of large-seeded plants was actually
240 associated with the decline among young of several radiated large-seeded species within
241 *Eugenia* / *Syzygium* (Myrtaceae), *Labourdonnaisia* (Sapotaceae) and *Diospyros* (Ebenaceae; Fig.A2).
242 Conversely, the higher decrease in phylogenetic diversity of medium-seeded plants at REU-ML came
243 from the high dominance of few medium-seeded Rubiaceae among young and from species of *Ocotea*
244 (Lauraceae), *Xylopia* (Annonaceae) and *Tambourissa* (Monimiaceae) showing very few individuals,
245 leading to low occurrence of Magnoliidae in the understory.

246 **Demographic status assessment of fleshy-fruited plants**

247 The median recruitment index of small- and medium-seeded FF trees was between three and five times
248 higher at MAU-BF (between 0.7 & 0.8, Fig.5) than at REU-ML (between 0.15 & 0.26). In other words,
249 half of small- and medium-seeded tree species had at least three times as many adults as juveniles at
250 REU-ML and at least three juveniles for every adult at MAU-BF. For large-seeded FF trees, the median
251 recruitment index was low at both sites (between 0.11 & 0.43), and the situation was paradoxically
252 better at REU-ML than at MAU-BF for young. If this result has to be tempered by the low diversity of
253 large-seeded trees at REU-ML, the collapse of the recruitment index observed in this group at MAU-BF
254 was particularly striking.

255 The median recruitment index of small- and medium-seeded FF shrubs was relatively similar at
256 MAU-BF and REU-ML (between 0.57 & 0.69 and between 0.45 & 0.58 for young and intermediate
257 cohorts, respectively, Fig.5). The median recruitment index of shrubs at REU-ML was thus four times
258 higher than that of trees with the same dispersal traits, despite the low number of shrub species.
259 Conversely, although not low at MAU-BF, the median recruitment index of small- and medium-seeded
260 shrubs was slightly lower than for juvenile trees. The few large-seeded shrubs in this study were only
261 encountered at MAU-BF and showed a particularly low median recruitment index (< 0.09). For details of
262 the results for each woody species, please report to the dataset (available at
263 <http://dx.doi.org/10.17632/r3x9hxt98s.1#file-2e500bfd-9b67-4bee-adac-e21e2d55f06c>).

264 **DISCUSSION**

265 Comparing three one-hectare plots of Mare Longue on Réunion (REU-ML) and Brise Fer on Mauritius
266 (MAU-BF), famous for being the land of the Dodo, is a unique opportunity to explore the consequences
267 of frugivores extinctions on the maintenance of diversity within tropical forests, as initiated by Terborgh
268 et al. (2008) in Peru. On Réunion, all large-bodied native frugivores went extinct 200 years ago and the
269 largest extant frugivore, the bulbul *Hypsipetes borbonicus*, weighs just over 50 g (Cheke and Hume,
270 2008). On Mauritius, all frugivores above 600 g (including the Dodo), are also extinct, but MAU-BF still
271 hosts a sister species of bulbul, but also a parakeet and more importantly a population of flying foxes
272 (*Pteropus niger*) which weighs up to 540 g (Fig.1 & Tab.A1). Our results support three assumed
273 differences between REU-ML considered as treatment (defaunated forest) and MAU-BF as pseudo-

274 control: (1) diversity is better preserved at MAU-BF than at REU-ML; (2) number of juveniles relative to
275 adults is larger at MAU-BF than at REU-ML, indicating stronger forest regeneration at MAU-BF;
276 (3) large-seeded plants have difficulties in regenerating in the understory at both sites. Our results
277 therefore indicate that the loss of frugivores likely has a severe impact on the regeneration of lowland
278 tropical forests in the Mascarenes. Hence, even protected forests on oceanic islands would be just as
279 much at risk as tropical forests in Neotropics (Terborgh et al., 2008), in Central Africa (Effiom et al.,
280 2013; Trolliet et al., 2019) and in Borneo (Harrison et al., 2013).

281 **Concern about strong diversity loss at REU-ML, encouraging situation at MAU-BF**

282 Overall woody species richness is about twice as high at MAU-BF than at REU-ML, and if the
283 associated factors are beyond the scope of this study, the maintenance of diversity across diameter
284 cohorts (from adult to young) at MAU-BF compared to REU-ML is noteworthy. Over a third (ca 35%) of
285 woody species do not have even a single sapling in the understory at REU-ML, compared to less than a
286 fifth (ca 18%) at MAU-BF, especially large-seeded FF plants. In accordance, the recruitment is notably
287 higher at MAU-BF than at REU-ML, except for large-seeded FF species, for which it is low at both sites.
288 Although REU-ML forest is considered as the best preserved tropical rainforest in Mascarene islands
289 (Strasberg, 1996), the extent of diversity loss in the understory and the poor recruitment of saplings for
290 most species, questions the long-term sustainability of native forests on Réunion (Albert et al., 2020b).
291 On the contrary, the important regeneration of numerous species at MAU-BF shows that weeding of
292 invasive alien plants since the mid-1990s (Lorence and Sussman, 1986; Monty et al., 2013), has
293 allowed this today promising recruitment (Fig.A3). Moreover, these observed differences between sites
294 are remarkable as invasive alien mammals as pigs, deer or macaques are present at MAU-BF and
295 absent at REU-ML (Florens, 2013). However, while the situation is encouraging at MAU-BF, our results
296 indicate that when immediate threats are removed, there are still ongoing problems, as most large-
297 seeded plants cannot recover despite conservation actions (Krivek et al., 2020). Therefore, it is likely
298 that MAU-BF forest will probably experience a shift in composition in the future with considerably less
299 large-seeded plants present in the canopy, as predicted elsewhere (Effiom et al., 2013; Harrison et al.,
300 2013; Terborgh et al., 2008). This would be all the more problematic as this loss would affect structural
301 tree species derived from the largest radiations, e.g. within *Eugenia*, *Syzygium* or *Diospyros* genera.

302 One might argue that we did not study community dynamics, and that caution must be taken
303 about the demographic status of plants, as few juveniles at a given time would not necessarily translate
304 in a population decline (Condit et al., 1998). Indeed, the non left-skewed diametric distribution might
305 reflect the normal depletion in the understory of a fast-growing species sampled at an unfavourable
306 time window. However, fast-growing species are very rare in the Mascarene flora (Cadet, 1977; Florens
307 et al., 2017b), and the lack of recruitment of numerous large-seeded plants that are theoretically
308 adapted to deeply shaded understory with strong competition (Leishman et al., 2000) suggests that
309 anomalies in static diameter distribution truly derive from failure in forest regeneration because of
310 strong dispersal limitation. Taking into account all dbh (and not only >1 cm) at MAU-BF shows, for
311 example, that the recruitment of large-seeded plants remains very low, contrary to small- and medium-
312 seeded FF plants which show a (very) large number of saplings with dbh < 1cm (Fig.A4). The difficulty
313 now lies in disentangling the mechanisms underlying this pattern of diversity, particularly since factors
314 other than the availability in frugivores, such as vegetation structure or other post-dispersal limitations,
315 fluctuate between the two sites.

316 **The likely crucial role of flying foxes in woody plant dispersal on Mauritius**

317 Diversity loss at both sites primarily involves FF plants that make the bulk of woody plants at both sites.
318 The disparities observed in the results between REU-ML and MAU-BF may be strongly influenced by a
319 more or less significant alteration of mutualism networks at archipelago scale (Schleuning et al., 2014).
320 Such a discrepancy in diversity maintenance and sapling recruitment of FF plants, except for large-

321 seeded ones, may be mainly due to the persistence of an important population of flying foxes on
322 Mauritius. Indeed, this bat is today the most important keystone species at MAU-BF due to the
323 extinction of all other large-bodied frugivores. Flying foxes are known to feed upon numerous native
324 species on Mauritius (Florens et al., 2017a; Nyhagen et al., 2005) and the role of Pteropodidae as
325 effective seed dispersers of small- and medium-seeded plants is well-known throughout the
326 Paleotropics (Bollen and Elsacker, 2002; McConkey and Drake, 2006; Muscarella and Fleming, 2007;
327 Oleksy et al., 2017; van Toor et al., 2019). Flying foxes probably ensure island-scale mobility for
328 numerous small-seeded native species that are sometimes absent from MAU-BF canopies (Oleksy et
329 al., 2019), ultimately allowing maintenance of local diversity. At REU-ML, the largest frugivore is the
330 Réunion bulbul, which occurs at a relative small density since the beginning of the 20th century (Cheke
331 and Hume, 2008), and might not ensure effective dispersal of many plant species even though it can
332 theoretically swallow their fruits (McConkey and O’Farrill, 2016). In fact, the good recruitment of FF
333 shrubs that the Réunion bulbul preferentially disperse (Albert, 2020) suggests that it still plays an
334 important role in the understory of old-growth forests. Similarly, the high increase in density of several
335 FF shrubs after weeding at MAU-BF (Baider and Florens, 2011) might be explained by consumption of
336 their fruits by the Mauritius bulbul. The population levels of this bird nevertheless remain relatively low
337 (< 600 individuals, Cheke and Hume, 2008).

338 Unlike other FF woody plants, large-seeded plants do not regenerate any better at MAU-BF
339 than they do at REU-ML on average. The weak increase of large-seeded tree recruitment at REU-ML
340 relative to smaller-seeded plants might be counterintuitive because of the extinction of all large native
341 frugivores since 1800, but it might also be an artifact. Indeed, several endangered large-fruited trees
342 might have already died out within permanent plots, e.g. *Sideroxylon majus* (Sapotaceae), *Hernandia*
343 *mascarensis* (Hernandiaceae), *Scolopia heterophylla* (Salicaceae). This would explain why there is so
344 low diversity among adults, and this would also lead to the overestimation of median recruitment of
345 extant species. Isolated large-seeded trees that still survive near the plots, hardly regenerate and might
346 go locally extinct, thus further impoverishing local diversity, and at the same time, increasing beta
347 diversity at larger spatial scale (Wandrag et al., 2017). Exceptions exist, as the large-seeded
348 *Labourdonnaisia calophylloides* that dominates plots at REU-ML and regenerates well, because
349 invasive rats may act as secondary seed dispersers (Shiels and Drake, 2011). However, while rats
350 could occasionally move large fruits, they are poor effective dispersers because their potential dispersal
351 distance is limited, but more importantly, they do destroy most seeds, and those hoarded are usually
352 clumped inside trees. Another possible contribution to this species regeneration are bird poachers that,
353 for centuries, have made glue from the fruit flesh (Cheke and Hume, 2008), reproducing to some extent
354 the behavior of extinct frugivores. Indeed, flesh persistence has a strong negative impact on the
355 regeneration of this species (Albert, 2020), as already demonstrated elsewhere in the Sapotaceae
356 family (Levi and Peres, 2013).

357 At MAU-BF, flying foxes feed upon large-seeded plants (Florens et al., 2017a), but fail to recruit
358 in the understory. Despite their relative large body mass allowing them to move large fruits, flying foxes
359 cannot swallow large seeds. This incapacity has two major implications as already demonstrated at
360 MAU-BF for the endangered endemic shrub *Syzygium mamillatum* (Myrtaceae): (1) seeds are
361 dispersed close from parents trees resulting in increased Janzen-Connell effects on seedlings and
362 saplings; (2) seeds are not cleaned properly, leading to the persistence of potential dormancy-inducing
363 or pathogen-attracting compounds, or increased attraction of introduced mammals (Hansen et al.,
364 2008). Anyhow, the effectiveness of large-seeded plant dispersal by extinct large-bodied frugivores was
365 probably much higher in the past. Entire lineages that had likely complementary functional roles in
366 frugivory went extinct at both sites: fruit pigeons including the dodo (Meehan et al., 2002; Whittaker and
367 Jones, 1994; Wotton and Kelly, 2011, 2012), parrots with the exception of the extant endangered
368 parakeet (Sebastián-González et al., 2019; Tella et al., 2020), skinks (Zuël et al., 2012) or giant
369 tortoises (Hansen et al., 2008). Their absence probably plays a major role in recruitment failure of
370 numerous large-seeded plants at both sites.

371 **Potential post-dispersal factors: seed predation, herbivory and light availability**

372 Recruitment failure in some large-seeded species may also be due to seed predation, such as in the
373 large tree *Canarium paniculatum* (Burseraceae) of which seeds are often predated by introduced rats at
374 MAU-BF (Auchoybur and Florens, 2005). However, seed predation by invasive mammals may be a
375 secondary cause of recruitment failure as already shown at MAU-BF (Hansen et al., 2008), on Réunion
376 (Albert, 2020) or in New Zealand (Wotton and Kelly, 2011). Furthermore, seedlings and saplings are
377 impacted by invasive herbivores, such as macaques or deers, as already reported by Thompson in the
378 late 19th century (Cheke and Hume, 2008). Large-seeded plants such as Sapotaceae or *Diospyros*
379 spp. may have specific vegetative traits which would make them more palatable than other plants
380 (Cheke and Hume, 2008). Again, herbivory might be a secondary limitation on large-seeded plants that
381 would primarily have difficulty establishing because of strong dispersal limitation.

382 One may argue that some species are shade-intolerant and have major difficulties in
383 regenerating under closed canopy where competition for light is strong. This is especially true for most
384 DF trees that include the smaller-seeded woody plants in the Mascarenes, as elsewhere (Leishman et
385 al., 2000). The fact that saplings of anemochorous long-lived pioneer species such as *Agarista*
386 *salicifolia* (Ericaceae), *Homalium paniculatum* (Salicaceae) or *Nuxia verticillata* (Stilbaceae) are absent
387 or very rare at REU-ML reinforces this hypothesis (Cadet, 1977; Strasberg, 1996), and contributes to
388 the observed decline in diversity. This influence of light availability is shown at MAU-BF where DF
389 plants such as *Homalium integrifolium* or *N. verticillata*, only regenerate well where canopy cover is the
390 lowest (Fig.A5). Canopy tree species with large, wind-dispersed seeds are absent in the Mascarenes, a
391 common feature of oceanic islands (Whittaker et al., 1997). Therefore, a gradual replacement of large-
392 seeded FF plants in the understory, as shown in continental tropical forests (Harrison et al., 2013;
393 Terborgh et al., 2008), is not possible.

394 Finally, native large-fruited light-demanding trees are usually absent in gaps at REU-ML
395 because they are not dispersed to such areas (Wandrag et al., 2017) and most native tree species do
396 not have a seedling/sapling bank prior to disturbance. These gaps are often occupied by alien FF small-
397 seeded plants that benefit from high levels of dispersal by small frugivores (Albert, 2020; Baret et al.,
398 2008; Mandon-Dalger et al., 2004), indicating the importance of seed dispersal in forest dynamics.

399 **RECOMMENDATIONS FOR CONSERVATION**

400 The protected tropical rainforest of REU-ML shows insufficient regeneration of most of its species
401 because of large-bodied frugivore loss, while that of MAU-BF shows an important recruitment of FF
402 plants, indicating their capacity of being effectively dispersed. There is consequently an urgent need to
403 protect flying foxes that face repeated mass culling on Mauritius (Florens and Baider, 2019) and to
404 promote the recolonization of larger forested areas by this species, which has now only a small
405 population on Réunion, after over 200 years of absence. While the ideal would be to rewild ecosystems
406 with large-gape frugivores as already undertaken in several South-West Indian Ocean islands (Griffiths
407 et al., 2011; Pedrono et al., 2013), direct sowing actions of large-seeded species should nevertheless
408 be considered to support their recruitment. Ultimately, we provide additional evidence of a bleak future
409 for diverse tropical rainforest if seed dispersal networks are not restored wherever necessary.

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414 **REFERENCES**

415 Albert, S., 2020. Rupture des interactions mutualistes plantes à fruits charnus-vertébrés frugivores, et

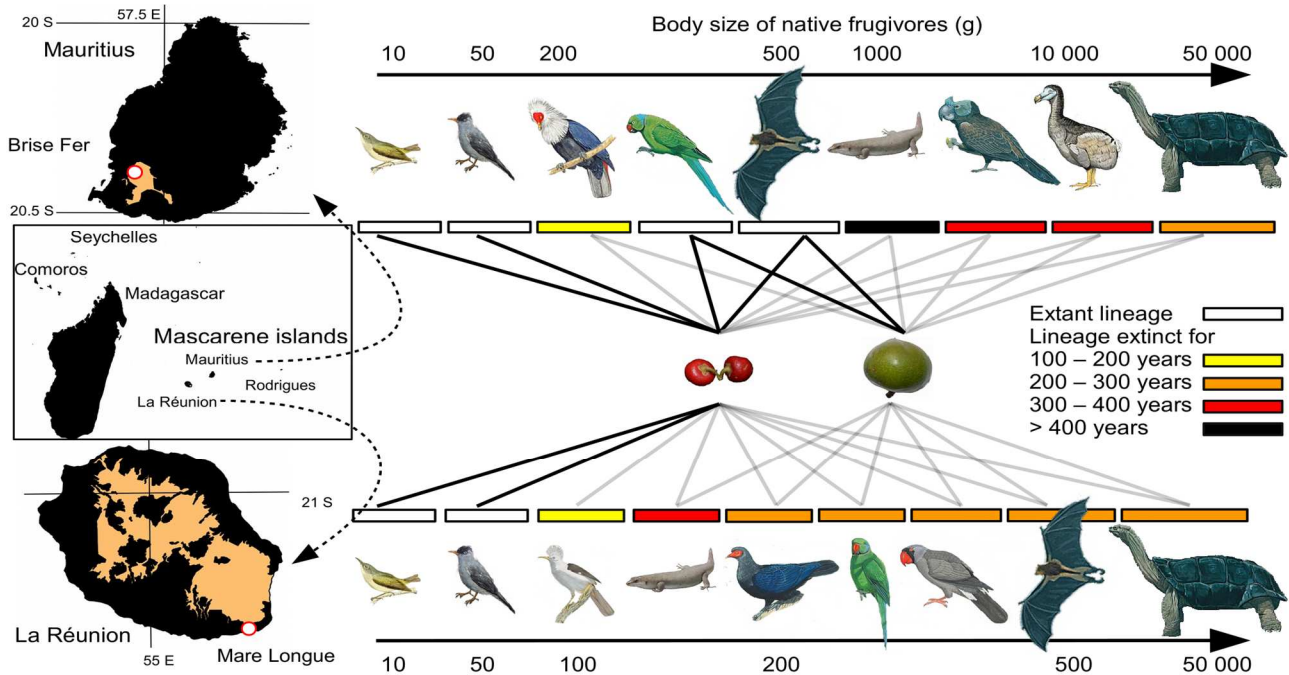
- 416 conséquences sur la régénération des forêts tropicales dans les Mascareignes (Doctoral dissertation).
417 Université de La Réunion.
- 418 Albert, S., Flores, O., Michon, L., Strasberg, D., 2020a. Dating young (<1000 yr) lava flow eruptions of
419 Piton de la Fournaise volcano from size distribution of long-lived pioneer trees. *J. Volcanol. Geotherm.*
420 *Res.* 401, 106974. <https://doi.org/10.1016/j.jvolgeores.2020.106974>
- 421 Albert, S., Flores, O., Rouget, M., Wilding, N., Strasberg, D., 2018. Why are woody plants fleshy-fruited
422 at low elevations? Evidence from a high-elevation oceanic island. *J. Veg. Sci.* 29, 847–858.
423 <https://doi.org/10.1111/jvs.12676>
- 424 Albert, S., Flores, O., Strasberg, D., 2020b. Collapse of dispersal trait diversity across a long-term
425 chronosequence reveals a strong negative impact of frugivore extinctions on forest resilience. *J. Ecol.*
426 108, 1386–1397. <https://doi.org/10.1111/1365-2745.13359>
- 427 Auchoybur, G., Florens, F.B.V., 2005. Threats to the maintenance of *Canarium paniculatum*
428 (Burseraceae) a Mauritian endemic canopy tree, in: *Frontiers in Tropical Biology and Conservation. The*
429 *Annual Meeting of the Association of Tropical Biology and Conservation, 24-28th July 2005, University*
430 *of Uberlândia, Brazil.* p. 164.
- 431 Baider, C., Florens, F.B.V., 2011. Control of invasive alien weeds averts imminent plant extinction. *Biol.*
432 *Invasions* 13, 2641–2646. <https://doi.org/10.1007/s10530-011-9980-3>
- 433 Baider, C., Florens, F.V., Baret, S., Beaver, K., Matatiken, D., Strasberg, D., Kueffer, C., others, 2010.
434 Status of plant conservation in oceanic islands of the Western Indian Ocean, in: *Proceedings of the 4th*
435 *Global Botanic Gardens Congress.* pp. 1–7.
- 436 Baret, S., Cournac, L., Thébaud, C., Edwards, P., Strasberg, D., 2008. Effects of canopy gap size on
437 recruitment and invasion of the non-indigenous *Rubus alceifolius* in lowland tropical rain forest on
438 Réunion. *J. Trop. Ecol.* 24, 337–345. <https://doi.org/10.1017/S0266467408004987>
- 439 Bollen, A., Elsacker, L.V., 2002. Feeding ecology of *Pteropus rufus* (Pteropodidae) in the littoral forest
440 of Sainte Luce, SE Madagascar. *Acta Chiropterologica* 4, 33–47. <https://doi.org/10.3161/001.004.0105>
- 441 Bosser, J., Cadet, T., Guého, J., Julien, H., Marais, W., 1976. *Flore des Mascareignes: Réunion,*
442 *Maurice, Rodrigues.* MSIRI, Mauritius, ORSTOM, IRD éditions, Paris and RBG, Kew, UK.
- 443 Cadet, T., 1977. *La végétation de l'île de la Réunion: étude phytoécologique et phytosociologique*
444 *(Doctoral dissertation).* Université d'Aix-Marseille.
- 445 Carpenter, J.K., Kelly, D., Moltchanova, E., O'Donnell, C.F.J., 2018. Introduction of mammalian seed
446 predators and the loss of an endemic flightless bird impair seed dispersal of the New Zealand tree
447 *Elaeocarpus dentatus*. *Ecol. Evol.* 8, 5992–6004. <https://doi.org/10.1002/ece3.4157>
- 448 Cheke, A.S., Hume, J.P., 2008. *Lost land of the dodo: an ecological history of Mauritius, Réunion &*
449 *Rodrigues.* Yale University Press, New Haven.
- 450 Comita, L.S., Queenborough, S.A., Murphy, S.J., Eck, J.L., Xu, K., Krishnadas, M., Beckman, N., Zhu,
451 Y., 2014. Testing predictions of the Janzen-Connell hypothesis: a meta-analysis of experimental
452 evidence for distance- and density-dependent seed and seedling survival. *J. Ecol.* 102, 845–856.
453 <https://doi.org/10.1111/1365-2745.12232>
- 454 Condit, R.S., Sukumar, R., Hubbell, S.P., Foster, R.B., 1998. Predicting population trends from size
455 distributions: a direct test in a tropical tree community. *Am. Nat.* 152, 495–509.
- 456 Connell, J.H., 1971. On the role of natural enemies in preventing competitive exclusion in some marine
457 animals and in rain forest trees, in: Den Boer, P.J., Gradwell, G.R. (Eds.), *Dynamics of Populations.* pp.
458 298–312.

- 459 Corlett, R., Primack, R., 2006. Tropical rainforests and the need for cross-continental comparisons.
460 Trends Ecol. Evol. 21, 104–110. <https://doi.org/10.1016/j.tree.2005.12.002>
- 461 Costa, J.B.P., Melo, F.P.L., Santos, B.A., Tabarelli, M., 2012. Reduced availability of large seeds
462 constrains Atlantic forest regeneration. Acta Oecologica 39, 61–66.
463 <https://doi.org/10.1016/j.actao.2011.12.002>
- 464 Dirzo, R., Young, H.S., Galetti, M., Ceballos, G., Isaac, N.J., Collen, B., 2014. Defaunation in the
465 Anthropocene. Science 345, 401–406.
- 466 Effiom, E.O., Nunez-Iturri, G., Smith, H.G., Ottosson, U., Olsson, O., 2013. Bushmeat hunting changes
467 regeneration of African rainforests. Proc. R. Soc. B Biol. Sci. 280, 20130246–20130246.
468 <https://doi.org/10.1098/rspb.2013.0246>
- 469 Faith, D.P., 1992. Conservation evaluation and phylogenetic diversity. Biol. Conserv. 61, 1–10.
470 [https://doi.org/10.1016/0006-3207\(92\)91201-3](https://doi.org/10.1016/0006-3207(92)91201-3)
- 471 Florens, F.B.V., Baider, C., 2019. Mass-culling of a threatened island flying fox species failed to
472 increase fruit growers' profits and revealed gaps to be addressed for effective conservation. J. Nat.
473 Conserv. 47, 58–64. <https://doi.org/10.1016/j.jnc.2018.11.008>
- 474 Florens, F.B.V., Baider, C., Marday, V., Martin, G.M.N., Zmanay, Z., Oleksy, R., Krivek, G., Vincenot,
475 C.E., Strasberg, D., Kingston, T., 2017a. Disproportionately large ecological role of a recently mass-
476 culled flying fox in native forests of an oceanic island. J. Nat. Conserv. 40, 85–93.
477 <https://doi.org/10.1016/j.jnc.2017.10.002>
- 478 Florens, F.B.V., Baider, C., Seegoolam, N.B., Zmanay, Z., Strasberg, D., 2017b. Long-term declines of
479 native trees in an oceanic island's tropical forests invaded by alien plants. Appl. Veg. Sci. 20, 94–105.
480 <https://doi.org/10.1111/avsc.12273>
- 481 Florens, V.F.B., 2013. Conservation in Mauritius and Rodrigues: challenges and achievements from
482 two ecologically devastated oceanic islands, in: Sodhi, N., Gibson, L., Raven, P.H. (Eds.), Conservation
483 Biology: Voices from the Tropics. John Wiley & Sons, Ltd., pp. 40–50.
- 484 Florens, V.F.B., Baider, C., Martin, G.M.N., Strasberg, D., 2012. Surviving 370 years of human impact:
485 what remains of tree diversity and structure of the lowland wet forests of oceanic island Mauritius?
486 Biodivers. Conserv. 21, 2139–2167. <https://doi.org/10.1007/s10531-012-0304-4>
- 487 Griffiths, C.J., Hansen, D.M., Jones, C.G., Zuël, N., Harris, S., 2011. Resurrecting extinct interactions
488 with extant substitutes. Curr. Biol. 21, 762–765.
- 489 Guimarães, P.R., Galetti, M., Jordano, P., 2008. Seed dispersal anachronisms: rethinking the fruits
490 extinct megafauna ate. PLoS ONE 3, e1745. <https://doi.org/10.1371/journal.pone.0001745>
- 491 Hansen, D.M., Galetti, M., 2009. The forgotten megafauna. Science 324, 42–43.
492 <https://doi.org/10.1126/science.1172393>
- 493 Hansen, D.M., Kaiser, C.N., Müller, C.B., 2008. Seed dispersal and establishment of endangered plants
494 on oceanic islands: The Janzen-Connell model, and the use of ecological analogues. PLoS ONE 3,
495 e2111. <https://doi.org/10.1371/journal.pone.0002111>
- 496 Harrison, R.D., Tan, S., Plotkin, J.B., Slik, F., Detto, M., Brenes, T., Itoh, A., Davies, S.J., 2013.
497 Consequences of defaunation for a tropical tree community. Ecol. Lett. 16, 687–694.
498 <https://doi.org/10.1111/ele.12102>
- 499 Heinen, J.H., van Loon, E.E., Hansen, D.M., Kissling, W.D., 2017. Extinction-driven changes in
500 frugivore communities on oceanic islands. Ecography 41, 1245–1255.

- 501 Howe, H.F., 2016. Making dispersal syndromes and networks useful in tropical conservation and
502 restoration. *Glob. Ecol. Conserv.* 6, 152–178. <https://doi.org/10.1016/j.gecco.2016.03.002>
- 503 Howe, H.F., Smallwood, J., 1982. Ecology of seed dispersal. *Annu. Rev. Ecol. Syst.* 13, 201–228.
- 504 Janzen, D.H., 1970. Herbivores and the number of tree species in tropical forests. *Am. Nat.* 104, 501–
505 528.
- 506 Kier, G., Kreft, H., Lee, T.M., Jetz, W., Ibisch, P.L., Nowicki, C., Mutke, J., Barthlott, W., 2009. A global
507 assessment of endemism and species richness across island and mainland regions. *Proc. Natl. Acad.*
508 *Sci.* 106, 9322–9327.
- 509 Krivek, G., Florens, F.B.V., Baider, C., Seegobin, V.O., Haugaasen, T., 2020. Invasive alien plant
510 control improves foraging habitat quality of a threatened island flying fox. *J. Nat. Conserv.* 54, 125805.
511 <https://doi.org/10.1016/j.jnc.2020.125805>
- 512 Laurance, W.F., Carolina Useche, D., Rendeiro, J., Kalka, M., Bradshaw, C.J.A., Sloan, S.P., Laurance,
513 S.G., Campbell, M., Abernethy, K., Alvarez, P., et al., 2012. Averting biodiversity collapse in tropical
514 forest protected areas. *Nature* 489, 290–294. <https://doi.org/10.1038/nature11318>
- 515 Leishman, M.R., Wright, I.J., Moles, A.T., Westoby, M., 2000. The evolutionary ecology of seed size, in:
516 Fenner, M. (Ed.), *Seeds: the ecology of regeneration in plant communities*. CABI, pp. 31–57.
- 517 Levi, T., Peres, C.A., 2013. Dispersal vacuum in the seedling recruitment of a primate-dispersed
518 Amazonian tree. *Biol. Conserv.* 163, 99–106. <https://doi.org/10.1016/j.biocon.2013.03.016>
- 519 Lorence, D.H., Sussman, R.W., 1986. Exotic species invasion into Mauritius wet forest remnants. *J.*
520 *Trop. Ecol.* 2, 147–162. <https://doi.org/10.1017/S0266467400000742>
- 521 Mandon-Dalger, I., Clergeau, P., Tassin, J., Rivière, J.-N., Gatti, S., 2004. Relationships between alien
522 plants and an alien bird species on Reunion Island. *J. Trop. Ecol.* 20, 635–642.
523 <https://doi.org/10.1017/S0266467404001774>
- 524 Matthews, T.J., Whittaker, R.J., 2015. REVIEW: On the species abundance distribution in applied
525 ecology and biodiversity management. *J. Appl. Ecol.* 52, 443–454. [https://doi.org/10.1111/1365-
526 2664.12380](https://doi.org/10.1111/1365-2664.12380)
- 527 McConkey, K.R., Drake, D.R., 2006. Flying foxes cease to function as seed dispersers long before they
528 become rare. *Ecology* 87, 271–276.
- 529 McConkey, K.R., O’Farrill, G., 2016. Loss of seed dispersal before the loss of seed dispersers. *Biol.*
530 *Conserv.* 201, 38–49. <https://doi.org/10.1016/j.biocon.2016.06.024>
- 531 Meehan, H.J., McConkey, K.R., Drake, D.R., 2002. Potential disruptions to seed dispersal mutualisms
532 in Tonga, Western Polynesia. *J. Biogeogr.* 29, 695–712.
- 533 Mittermeier, R.A., Hoffman, M., Pilgrim, J., Brooks, T., Mittermeier, C.G., Lamoreux, J.L., Da Fonseca,
534 G.A.B., Seligmann, P.A., Ford, H., 2004. Hotspots revisited. Earth’s biologically richest and most
535 endangered terrestrial ecoregions. Cemex, Mexico city.
- 536 Monty, M.L.F., Florens, F.B.V., Baider, C., 2013. Invasive alien plants elicit reduced production of
537 flowers and fruits in various native forest species on the tropical island of Mauritius (Mascarenes, Indian
538 Ocean). *Trop. Conserv. Sci.* 6, 35–49. <https://doi.org/10.1177/194008291300600107>
- 539 Muscarella, R., Fleming, T.H., 2007. The role of frugivorous bats in tropical forest succession. *Biol. Rev.*
540 82, 573–590. <https://doi.org/10.1111/j.1469-185X.2007.00026.x>
- 541 Nyhagen, D.F., Turnbull, S.D., Olesen, J.M., Jones, C.G., 2005. An investigation into the role of the

- 542 Mauritian flying fox, *Pteropus niger*, in forest regeneration. *Biol. Conserv.* 122, 491–497.
543 <https://doi.org/10.1016/j.biocon.2004.08.012>
- 544 Oleksy, R.Z., Ayady, C.L., Tatayah, V., Jones, C., Howey, P.W., Froidevaux, J.S.P., Racey, P.A.,
545 Jones, G., 2019. The movement ecology of the Mauritian flying fox (*Pteropus niger*): a long-term study
546 using solar-powered GSM/GPS tags. *Mov. Ecol.* 7, 1–12. <https://doi.org/10.1186/s40462-019-0156-6>
- 547 Oleksy, R.Z., Giuggioli, L., McKetterick, T.J., Racey, P.A., Jones, G., 2017. Flying foxes create
548 extensive seed shadows and enhance germination success of pioneer plant species in deforested
549 Madagascan landscapes. *PLOS ONE* 12, e0184023. <https://doi.org/10.1371/journal.pone.0184023>
- 550 Pavoine, S., 2020. *adiv*: An r package to analyse biodiversity in ecology. *Methods Ecol. Evol.* 11, 1106–
551 1112. <https://doi.org/10.1111/2041-210X.13430>
- 552 Pedrono, M., Griffiths, O.L., Clausen, A., Smith, L.L., Griffiths, C.J., Wilmé, L., Burney, D.A., 2013.
553 Using a surviving lineage of Madagascar’s vanished megafauna for ecological restoration. *Biol.*
554 *Conserv.* 159, 501–506. <https://doi.org/10.1016/j.biocon.2012.11.027>
- 555 Réchou, A., Flores, O., Jumeaux, G., Duflot, V., Bousquet, O., Pouppeville, C., Bonnardot, F., 2019.
556 Spatio-temporal variability of rainfall in a high tropical island: Patterns and large-scale drivers in
557 Réunion Island. *Q. J. R. Meteorol. Soc.* 145, 893–909. <https://doi.org/10.1002/qj.3485>
- 558 Safford, R.J., 1997. A survey of the occurrence of native vegetation remnants on Mauritius in 1993.
559 *Biol. Conserv.* 80, 181–188. [https://doi.org/10.1016/S0006-3207\(96\)00048-1](https://doi.org/10.1016/S0006-3207(96)00048-1)
- 560 Schleuning, M., Böhning-Gaese, K., Dehling, D.M., Burns, K.C., 2014. At a loss for birds: insularity
561 increases asymmetry in seed-dispersal networks. *Glob. Ecol. Biogeogr.* 23, 385–394.
562 <https://doi.org/10.1111/geb.12134>
- 563 Sebastián-González, E., Hiraldo, F., Blanco, G., Hernández-Brito, D., Romero-Vidal, P., Carrete, M.,
564 Gómez-Llanos, E., Pacífico, E.C., Díaz-Luque, J.A., Dénes, F.V., Tella, J.L., 2019. The extent,
565 frequency and ecological functions of food wasting by parrots. *Sci. Rep.* 9, 15280.
566 <https://doi.org/10.1038/s41598-019-51430-3>
- 567 Shiels, A.B., Drake, D.R., 2011. Are introduced rats (*Rattus rattus*) both seed predators and dispersers
568 in Hawaii? *Biol. Invasions* 13, 883–894. <https://doi.org/10.1007/s10530-010-9876-7>
- 569 Soltani, E., Baskin, C.C., Baskin, J.M., Heshmati, S., Mirfazeli, M.S., 2018. A meta-analysis of the
570 effects of frugivory (endozoochory) on seed germination: role of seed size and kind of dormancy. *Plant*
571 *Ecol.* 219, 1283–1294. <https://doi.org/10.1007/s11258-018-0878-3>
- 572 Strasberg, D., 1996. Diversity, size composition and spatial aggregation among trees on a 1-ha rain
573 forest plot at La Réunion. *Biodivers. Conserv.* 5, 825–840.
- 574 Strasberg, D., Rouget, M., Richardson, D.M., Baret, S., Dupont, J., Cowling, R.M., 2005. An
575 assessment of habitat diversity and transformation on La Réunion Island (Mascarene Islands, Indian
576 Ocean) as a basis for identifying broad-scale conservation priorities. *Biodivers. Conserv.* 14, 3015–
577 3032. <https://doi.org/10.1007/s10531-004-0258-2>
- 578 Tella, J.L., Hiraldo, F., Pacífico, E., Díaz-Luque, J.A., Dénes, F.V., Fontoura, F.M., Guedes, N., Blanco,
579 G., 2020. Conserving the diversity of ecological interactions: The role of two threatened macaw species
580 as legitimate dispersers of “megafaunal” fruits. *Diversity* 12, 45. <https://doi.org/10.3390/d12020045>
- 581 Terborgh, J., Nuñez-Iturri, G., Pitman, N.C.A., Valverde, F.H.C., Alvarez, P., Swamy, V., Pringle, E.G.,
582 Paine, C.E.T., 2008. Tree recruitment in an empty forest. *Ecology* 89, 1757–1768.
583 <https://doi.org/10.1890/07-0479.1>
- 584 Traveset, A., Rodríguez-Pérez, J., Pías, B., 2008. Seed trait changes in dispersers’ gut and

- 585 consequences for germination and seedling growth. *Ecology* 89, 95–106.
- 586 Trollet, F., Bauman, D., Forget, P.-M., Doucet, J.-L., Gillet, J.-F., Hambuckers, A., 2019. How
587 complementary are large frugivores for tree seedling recruitment? A case study in the Congo Basin. *J.*
588 *Trop. Ecol.* 35, 223–236. <https://doi.org/10.1017/S026646741900018X>
- 589 Turner, I.M., 2001. *The ecology of trees in the tropical rain forest*. Cambridge University Press,
590 Cambridge; New York.
- 591 van Toor, M.L., O'Mara, M.T., Abedi-Lartey, M., Wikelski, M., Fahr, J., Dechmann, D.K.N., 2019.
592 Linking colony size with quantitative estimates of ecosystem services of African fruit bats. *Curr. Biol.* 29,
593 R237–R238. <https://doi.org/10.1016/j.cub.2019.02.033>
- 594 Wandrag, E.M., Dunham, A.E., Duncan, R.P., Rogers, H.S., 2017. Seed dispersal increases local
595 species richness and reduces spatial turnover of tropical tree seedlings. *Proc. Natl. Acad. Sci.* 114,
596 10689–10694. <https://doi.org/10.1073/pnas.1709584114>
- 597 Wang, B.C., Smith, T.B., 2002. Closing the seed dispersal loop. *Trends Ecol. Evol.* 17, 379–386.
- 598 Whittaker, R.J., Jones, S.H., 1994. The role of frugivorous bats and birds in the rebuilding of a tropical
599 forest ecosystem, Krakatau, Indonesia. *J. Biogeogr.* 21, 245–258. <https://doi.org/10.2307/2845528>
- 600 Whittaker, R.J., Jones, S.H., Partomihardjo, T., 1997. The rebuilding of an isolated rain forest
601 assemblage: how disharmonic is the flora of Krakatau? *Biodivers. Conserv.* 6, 1671–1696.
- 602 Wotton, D.M., Kelly, D., 2012. Do larger frugivores move seeds further? Body size, seed dispersal
603 distance, and a case study of a large, sedentary pigeon. *J. Biogeogr.* 39, 1973–1983.
604 <https://doi.org/10.1111/jbi.12000>
- 605 Wotton, D.M., Kelly, D., 2011. Frugivore loss limits recruitment of large-seeded trees. *Proc. R. Soc. B*
606 *Biol. Sci.* 278, 3345–3354. <https://doi.org/10.1098/rspb.2011.0185>
- 607 Zanne, A.E., Tank, D.C., Cornwell, W.K., Eastman, J.M., Smith, S.A., FitzJohn, R.G., McGlenn, D.J.,
608 O'Meara, B.C., Moles, A.T., Reich, P.B., et al., 2013. Three keys to the radiation of angiosperms into
609 freezing environments. *Nature* 506, 89–92. <https://doi.org/10.1038/nature12872>
- 610 Zuël, N., Griffiths, C.J., Hector, A., Hansen, D.M., Jones, C.G., Albrecht, M., 2012. Ingestion by an
611 endemic frugivore enhances seed germination of endemic plant species but decreases seedling
612 survival of exotics. *J. Biogeogr.* 39, 2021–2030. <https://doi.org/10.1111/j.1365-2699.2012.02694.x>
- 613 **DATA ACCESSIBILITY**
- 614 Data associated with this study will be archived in the *Mendeley Data*. DOI:
615 <http://dx.doi.org/10.17632/r3x9hxt98s.1#file-2e500bfd-9b67-4bee-adac-e21e2d55f06c>



617
 618 **Fig.1** Overview of native frugivore guild at study sites. Réunion and Mauritius are the two main islands of the
 619 Mascarenes. Permanent plots of Mare Longue and Brise Fer (red circles) are located in the National Parks of
 620 Réunion and Mauritius, respectively (in orange). Réunion and Mauritius hosted a diverse and plethoric fauna
 621 of frugivores until the 17th century (from right to left, in descending order of body size): giant tortoises, dodo
 622 (Mauritius only), broad-billed parrot (Mauritius only), giant skink (Mauritius only), flying foxes, Mascarin parrot
 623 (Réunion only), parakeets, fruit pigeons, skinks, starlings (Réunion only), bulbuls and white-eyes. The
 624 extinction period of frugivore lineages is indicated by the rectangle colour. Extant and extinct frugivory
 625 interactions are given by black and grey lines, respectively. The red and green berries symbolize several
 626 hundred woody species with small and large fleshy fruits, large fleshy-fruited plants being theoretically not
 627 dispersible by small frugivores. Actually, the Mascarenes initially showed highly asymmetric dispersal
 628 networks with relatively few species of frugivores compared to the diversity of plants (see Schleuning et al.,
 629 2014). Illustrations: giant tortoises, dodo, flying foxes, parrots, fruit pigeons, starling by J. Hume; white-eyes
 630 by J.G. Keulemans; skinks by J. Noseworthy; bulbuls by D. Strasberg. For more details, see Appendix
 631 Tab.A1.

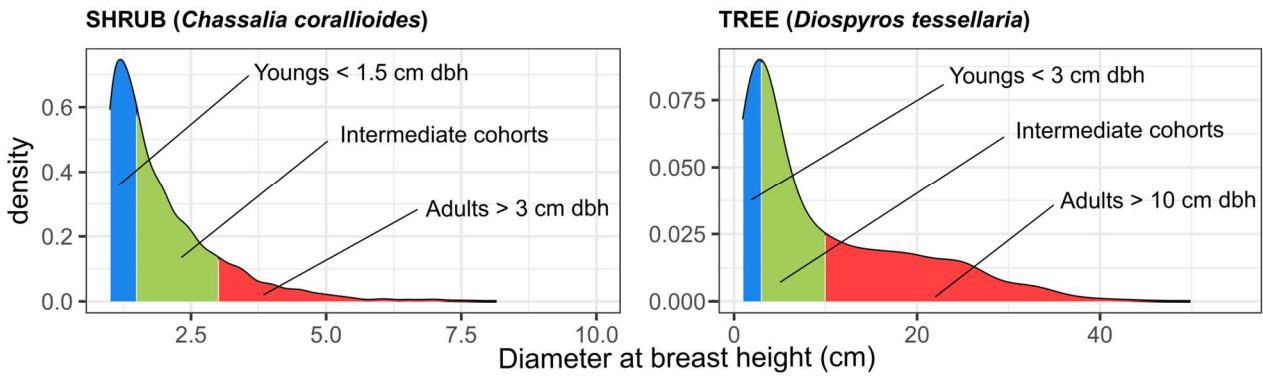
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633 **Tab.1** Descriptors of permanent plots at MAU-BF and REU-ML. Main environmental factors are shown:
 634 elevation (“asl”: above sea level) and annual precipitation (“Precip.”). Basal area and number of stems are
 635 also displayed depending on diameter cohorts. Basal area and number of stems account for all woody
 636 species.

Site	Plot	Elevation (m asl)	Precip. (mm)	Sampling year	Basal area dbh>1 cm (m ² .ha ⁻¹)	Basal area dbh>10 cm (m ² .ha ⁻¹)	Number of stems with dbh>1 cm	Number of stems with dbh>10 cm
Mauritius	BF1	560-600	2400	2011	41.6	34.8	6369	1026
	BF2	560-600	2400	2011	58.9	48.7	8003	1631
	BF3	560-600	2400	2010	65.2	58.4	5799	1463
Réunion	ML1	310-320	4540	2012	74.6	69.6	5807	984
	ML2	300-310	4520	2012	80.1	76.7	4580	935
	ML3	140-150	4375	2014	60.8	57.9	3098	814

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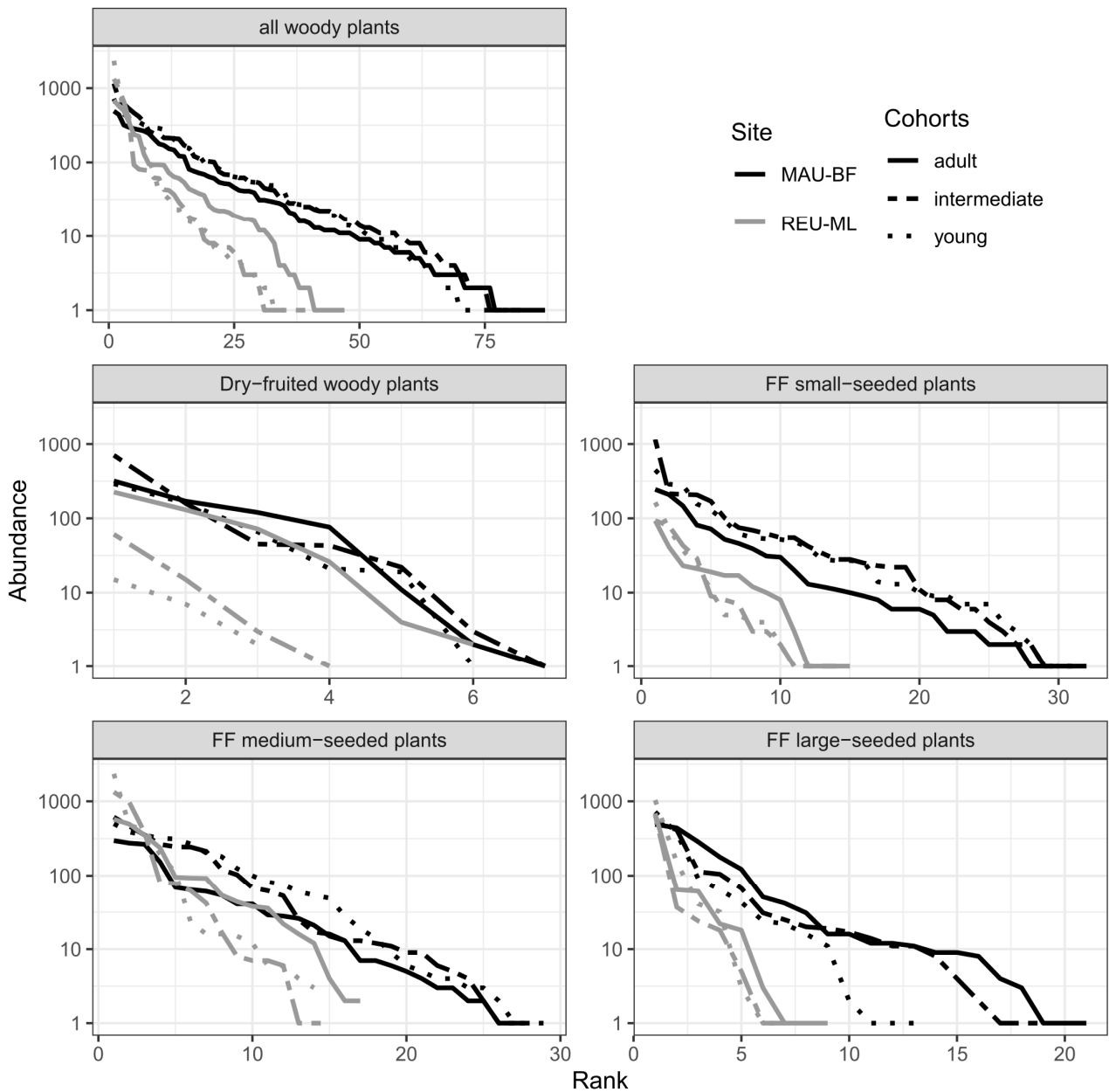
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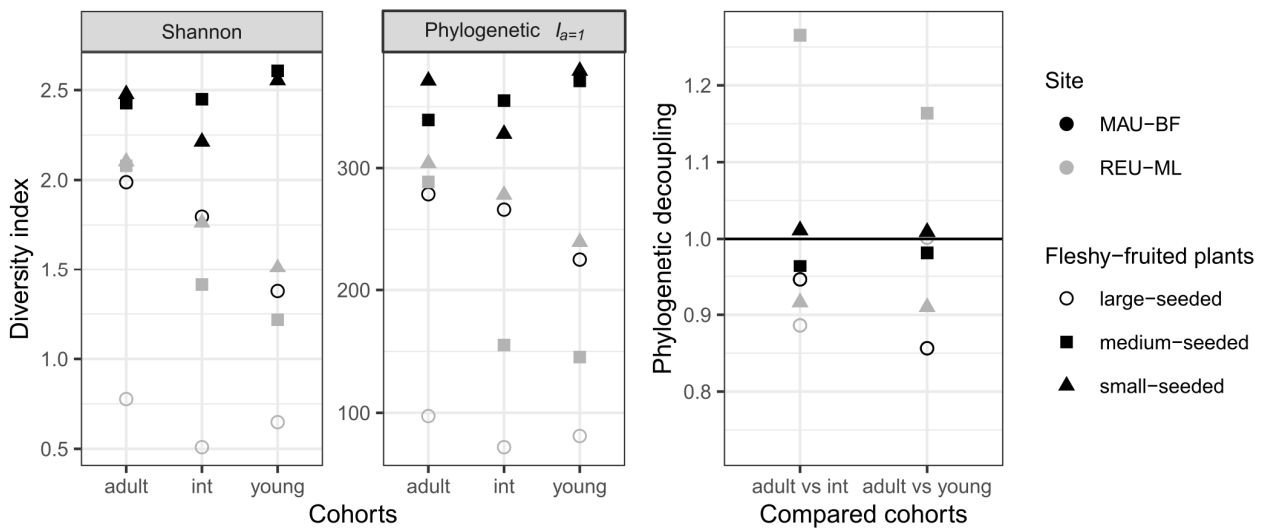
640 **Fig.2** Definition of diameter cohorts according to growth forms. Young cohorts of shrubs and trees show a
 641 diameter at breast height (dbh) of [1; 1.5[cm and [1; 3[cm (blue), respectively. Intermediate cohorts of
 642 shrubs and trees have a dbh of [1.5; 3[cm and [3; 10[cm (green), respectively. Adult cohorts of shrubs and
 643 trees have a dbh>3 cm and >10 cm (red), respectively. Two species with a left-skewed diametric distribution
 644 illustrate the assignment.

645

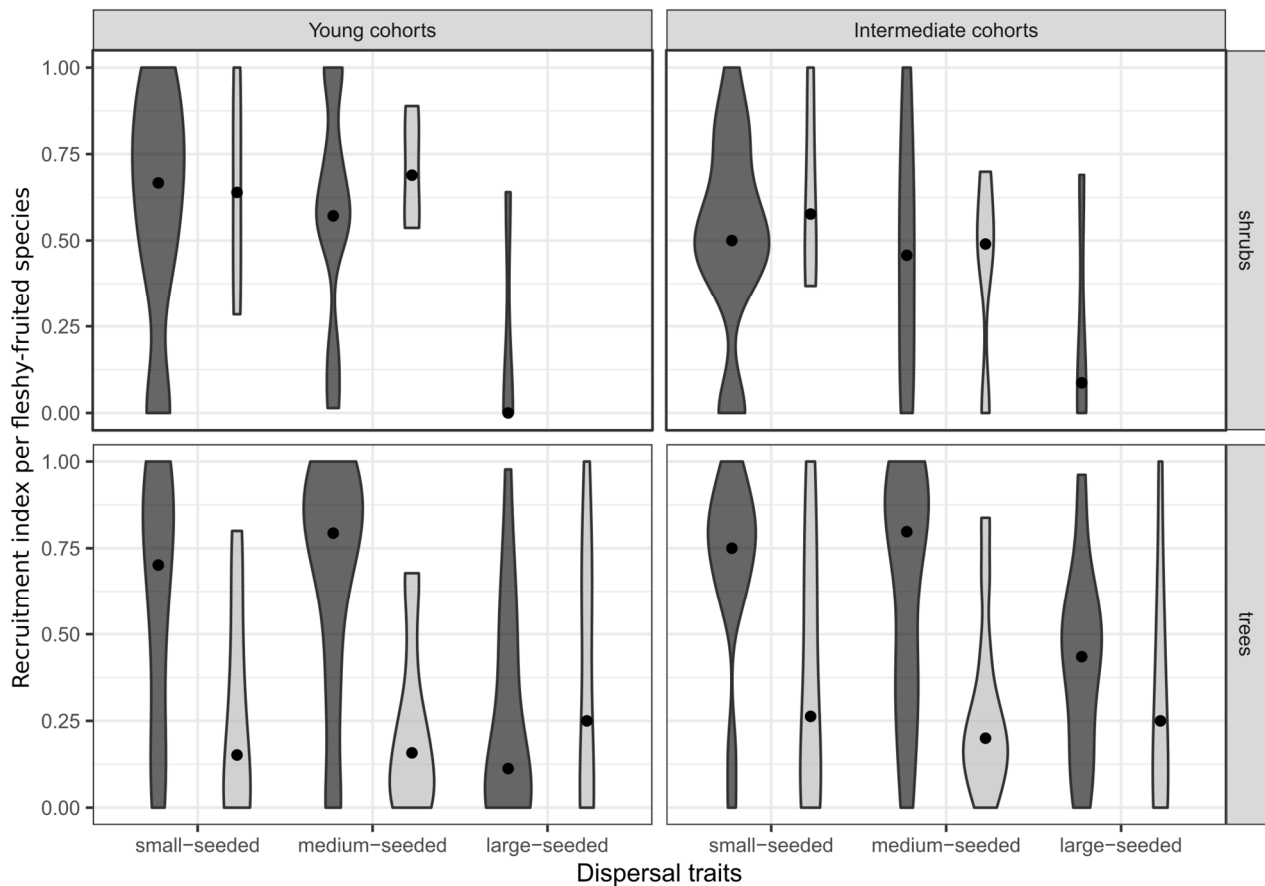


646 **Fig.3** Rank-abundance diagrams at MAU-BF (black) and REU-ML (grey) for woody species overall and
 647 according to dispersal traits: dry-fruited plants that were mainly anemochorous at both sites and fleshy-fruited
 648 (FF) small-seeded, medium-seeded and large-seeded plants of which mean seed length was [0.06; 0.6],
 649 [0.6; 1.5] and [1.5; 5.3] cm, respectively. Solid, dashed-dotted and dotted lines respectively displayed adult,
 650 intermediate and young cohorts. Note the different scale for the x-axis.

652



653
 654 **Fig.4** Changes in diversity indices of fleshy-fruited plant assemblages across diameter cohorts at MAU-BF
 655 (black) and REU-ML (grey). Left, Shannon diversity index; Center, phylogenetic diversity index $p_{l_{a=1}}$ of
 656 Pavoine (2020); Right, phylogenetic decoupling, e.g. for “adult vs young” ($p_{l_{a=1, adult}} / p_{l_{a=1, young}} / (I_{shannon, adult} /$
 657 $I_{shannon, young})$). Values of phylogenetic decoupling > 1 and < 1 indicated stronger and lower changes in
 658 phylogenetic diversity than in Shannon diversity across diameter cohorts, respectively. Circles, squares and
 659 triangles displayed large-, medium- and small-seeded fleshy-fruited plants, respectively. Note the different
 660 scale for the y-axis.



661
 662 **Fig.5** Violin plots of recruitment index of fleshy-fruited plants calculated as $n_{young} / (n_{young} + n_{adult})$ and $n_{int} / (n_{int}$
 663 $+ n_{adult})$ per species at MAU-BF (black) and REU-ML (grey) in relation to dispersal traits and growth forms. A
 664 value of 0.5 and 0.25 for young cohorts means that a species has as many youngs as adults and three times
 665 as many adults as youngs, respectively. Black dots display the median of observed distributions. Note that
 666 areas of violins are scaled proportionally to the number of species.