

17. Using relict species-area relationships to estimate the conservation value of reservoir islands to improve environmental impact assessments of dams

Authors

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17.1 Chapter overview

Large dams are emerging drivers of landscape-scale habitat fragmentation. When dams are constructed, particularly in low-lying and moderately undulating regions, they can cause extensive flooding and the consequential transformation of hilltops into islands. However, Environmental

25 Impact Assessments (EIAs; the process to assess and account for impacts of development on the
26 environment) do not explicitly consider these reservoir islands in calculations of habitat impacted by
27 dam construction. The Amazon Basin is low-lying and moderately undulating, so here, dam
28 construction can lead to the formation of vast reservoirs and the creation of extensive forest island
29 archipelagos. Reservoir islands are largely comprised of relict biological communities containing
30 species that were once part of the previously continuous habitat. Following isolation on reservoir
31 islands, these relict communities are subject to an ongoing extinction debt whereby species are lost
32 from islands over time. In this chapter we demonstrate how estimating the “conservation value” of
33 islands (CV: the proportion of relict continuous forest species found on islands) using relict species-
34 area relationships (R SARs), can be used in a simple area-of-impact correction tool to ensure that
35 environmental impacts to insular habitats are explicitly accounted-for in dam EIAs.

36

37 In our study we used data from eight taxonomic groups, including medium and large-sized
38 vertebrates, small mammals, understory birds, orchid bees, lizards, frogs, and adult and sapling trees
39 surveyed within the Balbina Hydroelectric Reservoir (BHR) archipelago and in nearby mainland
40 continuous forest, in Brazilian Amazonia. We show that all biological groups have positive R SARs
41 with the CV of islands increasing with island area. But a total of *ca.* 72,000 ha of insular habitat had
42 reduced CV (i.e. lower relict species richness compared to continuous forest) which equates to 60% of
43 aggregate island area. All terrestrial species are directly removed from the flooded area of reservoirs
44 and this is accounted for in EIAs; the same should hold for empty forests remaining above the water
45 line, such as reservoir islands lacking terrestrial species. We show that an additional 24% of the *ca.*
46 300,000 ha BHR water surface area should be included in area assessments for impacted terrestrial
47 habitat.

48

49 Taking the BHR as a case study, we demonstrate that using relict species-area relationships (R SARs)
50 and a simple area-of-impact correction tool that incorporates the conservation value of reservoir
51 islands, enables more accurate assessment of the total area of terrestrial habitat impacted by dam
52 development. In circumstances where inundation of highly biodiverse terrestrial habitats and the

53 creation of reservoir islands is unavoidable, using r SARs to assess the conservation value of reservoir
54 islands enables more accurate and dynamic assessment of the ecological impacts of dam construction.
55

56 17.2 Introduction

57

58 The Amazon, Mekong and Congo are our most biodiverse river basin systems globally. Dams both
59 constructed and planned for construction within these basins are emerging drivers of landscape- and
60 basin-scale habitat loss and fragmentation. This is because as reservoirs are filled, terrestrial habitat is
61 flooded and split apart to form smaller patches of remnant habitat “islands” within a water matrix
62 (Zarfl, Lumsdon & Tockner, 2015; Winemiller *et al.*, 2016; Moran *et al.*, 2018). The far-reaching
63 ecological impacts of river impoundment are of particular concern in highly biodiverse habitats such
64 as lowland tropical forests (Gibson, Wilman & Laurance, 2017). For example, the Amazon River
65 basin is threatened by prolific dam development, with more than 280 dams in operation or planned for
66 construction (Lees *et al.*, 2016; Latrubesse *et al.*, 2017).

67

68 Dam construction in moderately undulating lowland tropical regions results in the inundation of vast
69 areas of tropical forest, and wholesale changes to highly connected river systems due to disruption of
70 fisheries and flood pulses (Lees *et al.*, 2016). Alongside the direct loss of terrestrial habitats during
71 reservoir filling, archipelagos of forest islands are created from former hilltops. For example, the
72 construction of the Tucuruí Hydroelectric Reservoir (Brazil), the first major Amazonian dam (>15 m
73 dam height; ICOLD 2018) which began operation in 1984, led to the inundation of *ca.* 243,000 ha of
74 forest and the creation of some 2,200 reservoir islands. In 1987 the Uatumã River (a first-order
75 tributary of the Amazon River) was impounded by the Balbina Dam, which flooded *ca.* 300,000 ha of
76 continuous old-growth forest (Fearnside, 2016) and created > 3,500 islands (Benchimol & Peres,
77 2015a).

78

79 Dam-induced island archipelagos are true land-bridge islands, created simultaneously from previously
80 continuous habitat. Reservoir islands are surrounded by a uniform and inhospitable water matrix, and
81 are not buffered against the impacts of habitat fragmentation (i.e. edge-effects, such as wind damage)
82 that even a low-quality terrestrial habitat matrix may provide (Ewers & Didham, 2006). Thus, species
83 losses from habitat fragments embedded within a water matrix have been shown to be of a greater
84 magnitude than species losses from fragments within a terrestrial matrix (Watling & Donnelly, 2006).
85 For instance, when compared to continuous forest, forest islands in the Balbina archipelago have
86 greatly altered tree assemblage composition due to edge-related fires and wind-throws (Benchimol &
87 Peres, 2015a; Jones *et al.*, 2019) and severely disrupted animal-plant mutualistic networks that have
88 broken down due to landscape-scale habitat loss and fragmentation (Emer, Venticinque & Fonseca,
89 2013). Similar patterns of higher degrees of degradation of biological communities within a water
90 matrix have been demonstrated elsewhere for bats (Mendenhall *et al.*, 2014) and birds (Wolfe *et al.*,
91 2015).

92
93 Biological communities isolated on reservoir islands are largely comprised of relict species; i.e. those
94 species remaining from the previously continuous habitat (Watson, 2002). These relict communities
95 undergo a process of disassembly (whereby species are lost from islands) and experience ecological
96 ‘relaxation’ until new equilibrium communities are reached (Diamond, 1972; Gonzalez, 2000). Island
97 area — and hence the species-area relationship (SAR) — is a strong determinant of the rate of species
98 loss from reservoir islands through time, with larger islands able to retain more species and for longer
99 because conditions remain more closely aligned to those in continuous habitats. Smaller islands on the
100 other hand are at risk of rapid local extinctions because environmental conditions become much more
101 different compared to continuous habitat, alongside the fact that fewer species can be sustained in a
102 small area (Jones *et al.*, 2016). Thus, the “conservation value” (CV) of larger islands — in terms of
103 the number of relict species retained from the once continuous forest — is higher than that of smaller
104 islands.

105

106 There is widespread evidence of severe and long-term ecological impacts for relict biological
107 communities isolated on reservoir islands, particularly in highly biodiverse tropical regions (Jones *et*
108 *al.*, 2016). For instance, islands within the Chiew Larn reservoir (Thailand) have experienced near
109 total extinction of small mammals within 26 years of insularization, with 50% of species becoming
110 locally extinct in <14 years, precipitated by the appearance of an invasive species (Gibson *et al.*,
111 2013). In Lago Guri (Venezuela) local extinctions of top predators on islands caused trophic cascades,
112 severely impacting tree regeneration due to unchecked hyper-herbivory (Terborgh *et al.*, 2001, 2006);
113 while in the Brazilian Amazon, wholesale avifaunal erosion was induced from hundreds of local
114 extinctions on islands within the Tucuruí Hydroelectric Reservoir in 22-23 years of isolation history
115 (Bueno *et al.*, 2018). Thus, the local extinction of species from reservoir islands is an additional
116 ecological impact associated with dam-induced habitat fragmentation, which is not yet explicitly
117 included in the environmental impact assessments (EIAs) as a precondition of dam licensing (Gibson,
118 Wilman & Laurance, 2017; Ritter *et al.*, 2017).

119

120 The World Commission on Dams (WCD, 2000) outlined a comprehensive framework for assessing
121 the social and environmental impacts of dams, including recommendations to assess ecological
122 impacts at the basin-scale. Yet there is no binding signatory agreement for dam contractors,
123 developers, financiers or governments to forecast and mitigate dam-induced environmental impacts.
124 Moreover, the lack of any systematic and long-term monitoring of the efficacy of environmental
125 impact mitigation measures implemented, precludes the advancement of policies aimed towards
126 increasing the ecological sustainability of dams (Jones & Bull [accepted]; Moran *et al.* 2018). In
127 Brazil, *a priori* EIA has been required since 1986 to license any new major infrastructure project
128 (Ritter *et al.*, 2017). However, current EIAs are inadequate in terms of reporting the detrimental
129 effects of hydroelectric dams, on both natural and societal environments, including severe
130 underestimates of forest and biodiversity loss, greenhouse gas emissions, and displacement of
131 indigenous communities (Ritter *et al.*, 2017; Fearnside, 2016; Timpe & Kaplan, 2017).

132

133 In cases where environmental impact mitigation measures are proposed following EIAs of dams, they
134 may include ‘offsetting’ the area of land flooded (i.e. the reservoir water surface area) through for
135 example, strictly protecting an equivalent or greater area of comparable mainland habitat (Bull *et al.*,
136 2013). For instance, the Balbina Hydroelectric Reservoir (BHR; Brazilian Amazon) triggered the
137 creation of the *ca.* 940,000 ha Uatumã Biological Reserve, which covers a portion of the reservoir and
138 adjoining mainland continuous forest habitat. However, the majority of Amazonian dams do not have
139 an appropriate offset associated with them, and the long-term ecological impacts of tropical dam
140 construction remain poorly accounted-for (Sonter *et al.*, 2018; Latrubesse *et al.*, 2017).

141

142 Tropical forest habitats are of global importance for biodiversity — due to the enormous number of
143 species present — and ecosystem service provision including significant uptake and storage of
144 atmospheric CO₂ (Pan *et al.*, 2011). Yet these forests are under increasing threat from hydropower
145 expansion, causing forest loss and fragmentation through reservoir creation (Latrubesse *et al.*, 2017).

146 Therefore, understanding the fate of terrestrial taxa isolated on tropical reservoir islands is an
147 important component of the long-term environmental impact of tropical dams (Ritter *et al.*, 2017).

148 Despite the clear evidence of local species extinctions on reservoir islands, dam proponents have
149 suggested that reservoir islands can act as an effective means for biodiversity conservation as part of
150 environmental impact mitigation strategies (Trussart *et al.*, 2002). Moreover, if the area of impacted
151 terrestrial habitat is simply taken as the area of habitat flooded (i.e. the reservoir water surface area)
152 this may severely underestimate the long-term environmental impact of dams due to species on
153 reservoir islands being subject to local extinctions and a lasting extinction debt (Jones *et al.*, 2016).

154

155 Following the species-area relationship (SAR) islands sustain different numbers of relict species due
156 to their different areas. Thus, SARs of relict species on islands (i.e. the proportion of remnant species
157 from formally continuous habitat; _RSARs) present a powerful and accessible means for assessing the
158 ecological impact of insularization on terrestrial species. Depending on landscape topography and
159 reservoir levels, and hence whether many small or few large islands are created from the same amount
160 of flooding, the outcomes for relict species retention on islands may be contrasting, and require

161 different impact mitigation strategies (Ewers & Didham 2006; Jones & Bull [accepted]). Applying
162 relict species-area relationships (r SARs) to quantify relict species retention on reservoir islands
163 compared to continuous habitat therefore indicates the ‘conservation value’ of islands. By including
164 the area of reservoir islands that has reduced conservation value (i.e. the area of insular habitat that
165 has been impacted by river impoundment) into the calculations of impacted habitat, EIAs can be made
166 more accurate (Fig. 17.1). Additionally, SARs comprise one of the fundamental patterns in
167 biogeography and macroecology (MacArthur & Wilson, 1967; Lomolino, 2000) making them highly
168 applicable in conservation measures, including the design of protected areas and estimation of local
169 biological diversity (Matthews *et al.*, 2016).

170

171 In this study, we use the Balbina Hydroelectric Reservoir (BHR) as a case study to (1) demonstrate
172 how relict species-area relationships (r SARs) can be used to assess the conservation value of islands;
173 (2) present a novel area-of-impact correction tool, which incorporates the conservation value of
174 islands to estimate the *minimum* additional area of reservoir island habitat that must be considered in
175 EIAs to account for impacted insular habitat; and (3) based on combined data for eight vertebrate,
176 invertebrate and plant taxonomic groups (as a proxy for the whole biological community), determine
177 which individual groups can act as an indicator for the whole-community r SAR, to assist with
178 applying our conservation value and area-of-impact correction tool to other dam-induced
179 archipelagos.

180

181 17.3 Methods

182 17.3.1 Study area

183 We used the Balbina Hydroelectric Reservoir (BHR) archipelago, central Brazilian Amazonia (1°01 –
184 1°55 S; 60°290 – 59°28 W) as our study system. The *ca.* 300,000 ha BHR was formed following the
185 impoundment of the Uatumã River in 1987 (Fearnside, 2016). As the BHR filled, continuous old-
186 growth lowland tropical forest was inundated, transforming the moderately undulating landscape into
187 an archipelago of 3,546 islands, ranging in size from 0.3 to 4,878 ha (mean \pm SD = 33.4 \pm 156.3 ha).

188 Island area, isolation, and environmental disturbance at island edges shape the structure and
189 composition of these insular forests (Benchimol & Peres, 2015a; Jones *et al.*, 2017, 2019).

190

191 Within the BHR, our main study system comprised of a network of 72 focal islands (1.2 – 1,815 ha;
192 mean \pm SD = 132.5 \pm 318 ha) and three to four mainland continuous forest sites positioned across a
193 comparable elevation gradient (Fig. 17.2). Focal islands were selected based on cloudless
194 georeferenced Landsat ETM+ scenes from 2009 (230/061 and 231/061), were normally 1 km apart to
195 ensure spatial independence and were located at varying distances from the mainland. Using ArcGIS
196 (ESRI, 2012) we calculated island areas for all 3,546 islands within the BHR using Rapid-Eye high-
197 resolution (5 m pixel) imagery, covering 698,000 ha of the BHR landscape (Benchimol & Peres,
198 2015b).

199 17.3.2. Data collection

200 Using original published datasets, we compiled presence/absence data for eight biological groups
201 across focal islands and mainland continuous forest sites within the BHR system, surveyed within a 5-
202 year period (2011-2016; Table 17.1). Brief descriptions of the datasets follow, and the accompanying
203 references should be consulted for full details regarding species surveys.

204

205 *Medium and large vertebrates* — Medium and large-sized diurnal and nocturnal vertebrate species
206 (>100 g and amenable to line transect censuses, indirect sign surveys, armadillo surveys and camera
207 trapping) were surveyed between June 2011 and December 2012 (Benchimol & Peres, 2015b).

208 Primate, carnivore, xenarthran, ungulate, rodent, large bird, and tortoise species were recorded (Table
209 17.1).

210

211 *Small mammals* — Surveys were conducted using transects with live traps set at ground, understorey
212 and sub-canopy heights, and pitfall units connected by a drift fence. A total of 65,520 trap-nights
213 across 79 transects were completed from April to November in 2014 and 2015 (Table 17.1; Palmeirim
214 *et al.* 2018).

215

216 *Understorey birds* — Understorey birds were surveyed using mist nets deployed over 21,888 net-
217 hours between July and December in 2015 and 2016 (Table 17.1; Bueno & Peres 2019).

218

219 *Orchid bees* — Orchid (Euglossine) bees were surveyed using scent trap-arrays baited with cineole,
220 methyl salicylate, methyl cinnamate and vanillin in September 2012 and April 2013, with trap-arrays
221 exposed at each sampling site for three consecutive days (Table 17.1; Storck-Tonon & Peres 2017).

222

223 *Lizards* — Terrestrial diurnal lizard assemblages were sampled using pitfall traps connected by a drift
224 fence (Table 17.1; Palmeirim et al. 2017). Sampling was undertaken between April and November in
225 2014 and 2015, totalling 5,447 trap-days across 71 trapping plots that were each sampled for 16
226 consecutive days.

227

228 *Frogs* — Frog calls were recorded in riparian and non-riparian habitats using autonomous recording
229 units between July and December 2015 (Table 17.1; Bueno et al. [in review.]). Frog calls were
230 identified to species within a subset of 9,362 1-minute recordings totalling 156 hours.

231

232 *Adult and sapling trees* — Trees and arborescent palms ≥ 10 cm diameter at breast height (DBH;
233 hereafter referred to as adult trees) were surveyed in 2012 (Table 17.1; Benchimol & Peres 2015a).
234 Saplings of trees and arborescent palms that had the potential to reach ≥ 10 cm DBH (hereafter,
235 sapling trees) were surveyed in the same sites in 2014 (Table 17.1; Jones *et al.* 2019).

236

237 17.3.3 Data analysis

238 *Using α SARs to predict the conservation value of islands*

239 We compiled an ‘island x species’ and a ‘continuous forest x species’ presence/absence matrix for
240 each biological group, and for all groups combined as a proxy for the ‘whole community’. We pooled
241 species from all continuous forest sites to generate a representative ‘reference’ continuous forest

242 community. We took this conservative approach to avoid possible undersampling of highly biodiverse
243 habitats and to avoid bias towards very low island CVs: by using the maximum surveyed area of
244 continuous forest available as a ‘reference’ community, there was the best chance of detecting shared
245 species between islands and continuous forest. We excluded species from our analyses that were
246 present on islands but not in continuous forest because the focus of our study was the capacity of
247 islands to retain relict species from formerly continuous habitats, rather than their ability to support
248 newly-immigrated disturbance-adapted species (Ewers & Didham, 2006). To estimate the
249 ‘conservation value’ (CV) of each island, we calculated the proportion of continuous forest species
250 present on the island (i.e. relict species). Accordingly, an island harbouring all species found in
251 continuous forest sites would have a maximum CV of one, whereas an island harbouring none of
252 those species would have the minimum CV of zero.

253

254 For each biological group and the whole community, we generated species-area curves for relict
255 species (r SARs) for focal islands by modelling CV with \log_{10} island area ($ISLAND_{AREA}$; ha) using a
256 generalised linear model (GLM) with a quasibinomial error structure, and used these GLMs to predict
257 the CV of unsurveyed islands across the entire BHR archipelago.

258

259 *Estimating the area of islands with reduced conservation value*

260 We estimated the area of each of the 3,546 BHR islands with reduced CV (i.e. the area of island
261 habitat impacted by insularization; $IMPACTED_{ISLAND}$; ha) using the following equation:

262

$$263 \quad IMPACTED_{ISLAND} = ISLAND_{AREA} - (ISLAND_{AREA} * CV)$$

264

265 For each biological group and the whole community, we summed $IMPACTED_{ISLAND}$ for all 3,546 BHR
266 islands to give $IMPACTED_{TOTAL}$. Taking the reservoir area of 300,000 ha (Fearnside, 2016), we then
267 used the $IMPACTED_{TOTAL}$ estimates to ascertain the *minimum* percentage of additional reservoir water
268 surface area — on top of reservoir water surface area alone — that should be included in EIAs to

269 account for impacted island habitat. All analyses were performed using R (version 3.5.1; R Core
270 Team 2018).

271

272 *Assessing which biological groups can act as a proxy for the whole community*

273 We evaluated each biological group considering (1) our expert opinion, (2) survey feasibility, (3)
274 alignment of individual biological group $rSARs$ with that of the proxy ‘whole community’, and (4)
275 the percentage additional reservoir water surface area required to account for reduced biodiversity CV
276 on islands, to assess which groups can act as an indicator for relict species retention patterns, in order
277 to aid the application of our method to other landscapes affected by dam-induced habitat
278 fragmentation (Gardner *et al.*, 2008).

279

280 17.4 Results

281

282 *Using $rSARs$ to predict the conservation value of islands*

283 All taxonomic groups demonstrated positive $rSARs$, with the proportion of continuous forest species
284 retained on islands — i.e. the conservation value (CV) of islands — increasing with island area (Fig.
285 17.3). Considering the proxy ‘whole community’, the CV of islands ranged from 0.06 (in a 1.4-ha
286 island) to 0.66 (in a 4,878-ha island; mean CV \pm SD: 0.22 ± 0.07 ; Table 17.2), and rapidly increased
287 as a function of island area (slope = 0.738; Fig. 17.3). Across the entire range of island sizes within
288 the BHR (0.3-4,878 ha; mean \pm SD = 33.4 ± 156.3 ha) the maximum CV of an island was 0.96 (4,878
289 ha; mid- to large-sized vertebrates; Table 17.2). Conversely, five small islands (1.16-11.5 ha) had zero
290 CV for either mid- to large-bodied vertebrates or frogs.

291

292 *Estimating the area of islands with reduced conservation value*

293 The aggregate area of all islands within the BHR is 118,268 ha. Of that total area of insular habitat,
294 the total area exhibiting reduced CV ranged from 47,031 ha (40%) for medium and large vertebrates,
295 to 94,123 ha (80%) for understory birds (Table 17.2). When the ‘whole community’ was considered,

296 *ca.* 72,000 ha (*ca.* 60%) of island habitat had reduced CV. If the overall reservoir water surface area
297 (*ca.* 300,000 ha for the BHR; Fearnside 2016) is used in area calculations of impacted terrestrial
298 habitat, then this *ca.* 72,000 ha of impacted island habitat represents an additional 24% of the
299 reservoir water surface area that should be included in EIAs (Table 17.2). In other words, the total
300 impacted area of terrestrial habitat according to our method would be 372,000 ha (300,000 + 72,000
301 ha). However, certain biological groups — namely understory birds, frogs, and sapling trees — were
302 impacted over an additional *ca.* 25 to 31% of the *ca.* 300,000 ha of reservoir area (Table 17.2).

303

304 *Assessing which taxonomic groups can act as a proxy for the whole community*

305 The magnitude of impact (i.e. $rSAR$ slope) for all biological groups combined — the proxy ‘whole
306 community’ — was 0.738 (Fig. 17.4) and an additional 24% of the reservoir water surface area would
307 need to be included in EIAs to account for the reduced CV of reservoir islands at the community-
308 level. Based on (1) our expert opinion, (2) survey feasibility, (3) alignment of individual biological
309 group $rSARs$ with that of the proxy ‘whole community’, and (4) the percentage of additional reservoir
310 water surface area required to account for reduced biodiversity CV on islands, our results suggest that
311 plants (both adult and sapling trees; slope = 0.598, 23.3%; and 0.796, 25.3% respectively) and
312 understory birds (slope = 0.853, 31.4%) can be used as appropriate indicator groups for the response
313 of the proxy ‘whole community’ (Fig. 17.3; Fig. 17.4; Table 17.2; Gardner *et al.* 2008; Watson 2002).

314

315 17.5 Discussion

316 We demonstrate that $rSARs$ can be used effectively to estimate the conservation value of reservoir
317 islands to improve the accuracy of EIAs of major hydroelectric dams using a simple area-of-impact
318 correction tool. Across eight taxonomic groups within the Balbina Hydroelectric Reservoir (BHR)
319 landscape, the conservation value of reservoir islands co-varied tightly with island area, with
320 conservation value sharply declining with decreasing island area. Incorporating the conservation value
321 of islands into assessments of impacted terrestrial habitat indicated that an additional 24% of the
322 reservoir water surface area should be included in EIAs to account for impacted insular habitats.

323 Based on the degree to which r SARs of any given biological group tracked that of the proxy ‘whole
324 community’, we suggest that adult and sapling trees, as well as understorey birds, can serve as
325 appropriate indicator groups for community-level responses to insularization across the BHR.

326

327 *Reservoir islands have reduced conservation value*

328 All islands in the BHR had reduced conservation value compared to mainland continuous forest, with
329 smaller islands exhibiting the greatest reduction in conservation value (i.e. the lowest proportion of
330 continuous forest species retained). Previous studies of biological communities isolated on reservoir
331 islands have consistently shown that those isolated on small islands (<10 ha) are the most vulnerable
332 to local extinctions (Benchimol & Peres, 2015b; Terborgh *et al.*, 2001; Watson, 2002). However,
333 disturbance to forest structure, including wind damage, fire and desiccation at fragment edges, can
334 penetrate into remnant forests to varying extents and have been shown to affect forest biomass >1.5
335 km from fragment edges (Chaplin-Kramer *et al.*, 2015). Thus, species isolated on much larger islands
336 are still likely to be impacted by insularization. Indeed, our modelling exercise suggested that even the
337 largest island in the BHR (4,878 ha) did not support a full complement of species found in mainland
338 continuous forest.

339

340 *Estimating additional island habitat area to be included in Environmental Impact Assessments*

341 Incorporating the conservation value of islands into our area-of-impact correction tool revealed that of
342 the total aggregate island area (1 18,267 ha, n= 3,546), *ca.* 60% retained reduced CV when the entire
343 biological community was considered. Thus, in our case study, the equivalent of at least 24% of the
344 BHR reservoir water surface area would need to be added on top of the reservoir water surface area
345 alone, to account for the area of insular habitat with reduced conservation value. The surface area of
346 the BHR is *ca.* 300,000 ha (Fearnside, 2016) and therefore an additional *ca.* 72,000 ha (i.e. 372,000 ha
347 in total) would need to be incorporated into assessments of terrestrial habitat impacted by dam
348 development.

349

350 Using r SARs to estimate the conservation value of reservoir islands allows the impact of
351 insularization on relict continuous habitat species to be assessed and accounted-for in EIAs. r SARs
352 avoid artificial inflation of island conservation value by focussing on relict continuous forest species
353 and excluding newly-immigrated disturbance-adapted species. Our r SAR and island conservation
354 value approach drives the EIA process forward beyond simply assuming that the reservoir water
355 surface area equates to the total area of terrestrial habitat impacted (WCD, 2000; Ritter *et al.*, 2017).
356 EIAs do not explicitly consider reservoir island habitat in assessments of terrestrial land impacted by
357 dam construction, neglect biodiversity losses from islands and any other impacted mainland habitat,
358 and fall short of assessing many of the other direct and indirect ecological impacts of dam
359 construction in mega-diverse tropical regions (Lees *et al.*, 2016; Feamside, 2016; Timpe & Kaplan,
360 2017). We show that species loss from islands induced by flooding moderately undulating terrain,
361 such as in the Amazon Basin, is a significant additional environmental impact to be considered in the
362 EIA process (Jones & Bull [accepted]; Ritter *et al.* 2017).

363

364 However, we caution that the BHR archipelago likely represents the best-case-scenario for remnant
365 insular habitat and relict species retention, due to the fact that almost half of BHR islands and
366 mainland continuous forest to the east of the former Uatumã River bank are under strict protection
367 from the Uatumã Biological Reserve. Thus, there is minimal impact from other drivers of species loss
368 such as subsistence farming, logging and hunting, which many other dams may be subject to and
369 where species loss from islands may be far greater (Peres, 2001). Related to this, we also highlight
370 that depending on the condition of the ‘baseline’ habitat from which r SARs are generated, the
371 outcomes for island conservation values may be very different and may shift over time (Maron *et al.*,
372 2018). Moreover, our approach provides a snapshot of the habitat area correction needed to account
373 for reduced conservation value of islands *ca.* 30 years post-isolation. We therefore caution that the full
374 effects of insularization on relict species is yet to be realised because our method does not include a
375 means of incorporating ongoing extinction debts on islands, which may continue beyond 90 years of
376 island isolation (Jones *et al.*, 2016).

377

378 We focus on the role of island area in determining the degree of relict species retention, as this metric
379 is likely the most accessible for dam developers and decision-makers during the EIA process, because
380 the shape of the reservoir can be predicted using GIS and terrain maps with different water level
381 scenarios. Yet other landscape metrics, such as isolation distance and the degree of habitat
382 connectivity may also play important roles in shaping $rSARs$ on islands following insularization
383 (Fahrig, 2013; Palmeirim *et al.*, 2018; Storck-Tonon & Peres, 2017). When focussing on individual
384 taxonomic groups, a more nuanced relationship may emerge regarding the ability of habitat fragments
385 to retain species found in continuous habitats as demonstrated by the variation in $rSARs$ among the
386 eight biological groups in our study (Laurance *et al.*, 2011; Michalski, Nishi & Peres, 2007).

387

388 *Trees and understorey birds are appropriate indicators for whole community responses to*
389 *insularization*

390 The response of trees and understorey birds to insularization is relatively well documented (see e.g.
391 Aurélio-Silva *et al.* 2016; Benchimol & Peres, 2015; Bueno *et al.* 2018; Bueno & Peres 2019; Jones *et*
392 *al.*, 2019; Terborgh *et al.* 1997). We found that the $rSARs$ for both adult and sapling trees and
393 understorey birds aligned well with the community-level $rSAR$. In addition, tree and bird
394 communities play vital roles in the maintenance of critical ecological processes and services of forest
395 ecosystems, such as carbon storage, pollination, seed dispersal and pest control (Anderson-Teixeira *et*
396 *al.*, 2016; Bregman, Sekercioglu & Tobias, 2014), are key groups in long-term monitoring
397 programmes (Laurance *et al.*, 2011) and species identities can be double-checked *a posteriori* using
398 photographs and vegetative specimens, which reduces the potential for observer bias. Thus, we
399 recommend that at a minimum trees or understorey birds should be surveyed across insular habitats
400 when attempting to assess their conservation value.

401

402 Importantly, the conservation value of islands for understorey birds was lower than for the proxy
403 ‘whole community’. Therefore, using a general whole-community $rSAR$ to estimate conservation
404 value leads to underestimating the impacts for individual biological groups, and over-accounting for

405 others, such as orchid bees in this study. There may also be differences in r SARs within biological
406 groups depending on functional traits; e.g. for generalist vs specialist species. For example, in another
407 study of bird communities in an Amazonian dam-induced archipelago, forest specialist birds showed a
408 more severe decline in island occupancy as island area decreased (Bueno *et al.*, 2018), a finding
409 echoed in other groups in the BHR including lizards and small mammals (Palmeirim *et al.*, 2018;
410 Palmeirim, Vieira & Peres, 2017). Our approach therefore avoids the confounding effect of replacing
411 habitat specialist species with generalists, by focusing on relict species from continuous mainland
412 habitats: including newly immigrated disturbance-adapted species would overestimate the
413 conservation value of forest islands.

414

415 *Conclusions*

416 Using relict species-area relationships (r SARs) and a simple area-of-impact correction tool, which
417 incorporates the conservation value of reservoir islands, enables more accurate assessment of the total
418 area of terrestrial habitat impacted by dam development. Employing our method improves the
419 accuracy of EIAs of landscape-scale habitat fragmentation caused by dam development.

420

421 17.6 References

422

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