

1 **The effects of reduced impact logging and logging intensity on stand**
2 **damage, biomass loss and tree species richness in tropical forests: a**
3 **meta-analysis**
4

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16

17 **Abstract**

18

19 **Background.** At least a fifth of tropical forests have been logged in the recent past. This practice is
20 an important source of timber but there are concerns about its long-sustainability and impacts on
21 biodiversity and carbon storage. However, there is wide variation in the impacts of logging, making
22 generalisation, and thus policy implementation, difficult. Recent syntheses of animal biodiversity
23 have indicated that differences in logging intensity – the volume of wood removed per hectare –
24 may help explain some of these differences. In addition there have been suggestions that reduced
25 impact logging (RIL) may reduce some of the negative effects of logging.

26 **Methods.** We aimed to test these hypotheses using meta-analyses to explore differences in the
27 impacts of logging on (1) residual tree damage, (2) aboveground biomass and (3) tree species
28 richness.

29 **Results.** Our results indicate that RIL may reduce residual tree damage when compared to
30 conventional methods, but that at higher logging intensities this effect is negated. Changes in
31 aboveground biomass were negatively related to logging intensity, but any effect of RIL was
32 obscured by it being carried out at relatively low intensities. Tree richness appeared to initially
33 increase at low intensities but was reduced at higher intensities.

34 **Discussion.** Our results give only weak support to the hypothesis that RIL reduces the negative
35 impacts of logging on tree damage, and do not support suggestions that RIL reduces loss of biomass
36 or species richness. However, we do not think this is because there is no difference between the
37 impacts of RIL and conventional logging but rather that better evidence is needed to assess these
38 differences. We suggest that studies that take account of plot-level differences in logging intensity
39 are likely to provide a solution to this knowledge gap.

40

41 Introduction

42 Over 400 million hectares of tropical forest are designated as logging concessions (Asner et
43 al., 2009), making selective logging – the removal of selected trees from a stand – one of the most
44 widespread human disturbances in tropical forests (Asner et al., 2009). Tropical logging produces
45 approximately one eighth of global timber (Blaser et al., 2011) and is an important contributor to
46 many local and national economies. However, logging can have negative impacts on biodiversity
47 (Berry et al., 2010) and lead to increased carbon emissions (Bryan and Shearman, 2010; Nepstad et
48 al., 1999). Current practices can endanger the long-term sustainability of timber production
49 (Gourlet-fleury et al., 2013) and there have been suggestions that we may be approaching ‘peak
50 timber’ in the tropics (Shearman et al., 2012).

51 Logging impacts are driven by its effects on mortality and recruitment of trees and thus
52 forest structure. Mortality of large trees in selectively logged forests is high compared to
53 undisturbed forests, since large trees are usually those with highest timber value and are therefore
54 more likely to be harvested (Lindenmayer et al., 2013) However, mortality of smaller non-timber
55 trees is also often increased in logged forests when compared to undisturbed forests, since
56 harvesting and transportation of logs can result in damage to non-target trees (Picard et al., 2012).
57 These increases in mortality result in a reduction in biomass (Putz et al., 2012) and can also result in
58 changes in tree community composition, for example when the species recruited are not the same as
59 those that have died, or when recruitment cannot keep pace with mortality (Baraloto et al., 2012;
60 Gourlet-Fleury et al., 2013a). However, logging in tropical forests does not have a uniform effect on
61 forest structure (Burivalova et al. 2014), and there is substantial variation in logging impacts on tree
62 damage, biomass and biodiversity. For this reason, there is a need for meta-analyses of logging
63 impacts, in order for general responses to be identified.

64 Previous meta-analyses of the impacts of selective logging of tropical forest have indicated
65 that biomass losses following logging can be as low as 4% or as high as 66%, while changes in tree
66 species richness can vary between -53% to +27% of the values found in unlogged forests (Putz et
67 al., 2012). However, the potential causes of this variation have not been explored in previous
68 reviews (Clark and Covey, 2012; Putz et al., 2012). As logging intensity (the volume of wood
69 extracted per hectare) increases, so does the number of trees harvested and the proportion of
70 residual trees that are damaged (Picard et al., 2012). A higher logging intensity is likely to lead to
71 greater reductions in above-ground biomass because of the greater removal of trees and increased
72 mortality of non-target trees (Mazzei et al., 2010; Sist et al., 1998). The relationship between
73 logging intensity and species richness appears to be more complex, with the potential colonisation
74 of generalist species leading to maintenance or an increase in species richness with low intensity

75 logging and a reduction of richness at higher intensities. For example, a humped relationship
76 between logging intensity and species richness has been identified for birds, but changes in richness
77 for other vertebrates tend to be linear reductions (Burivalova et al., 2014).

78 In addition to logging intensity, a recent meta-analysis highlighted the potential of
79 differences in the methods used for logging in determining impacts. Bicknell et al. (2014) showed
80 that for similar logging intensities reduced impact logging (RIL) had less negative effects on animal
81 populations than conventional logging methods. RIL involves cutting lianas prior to logging, felling
82 trees in directions selected to cause least impact to surrounding forest, and limiting road
83 construction (Pinard and Putz 1996). Employment of these methods may reduce residual tree
84 damage compared to conventional logging, while reducing impacts on tree biomass and biodiversity
85 (Gullison and Hardner, 1993; Pinard and Putz, 1996). Some studies have suggested that RIL can be
86 carried out at similar intensities to those of conventional logging while causing less damage to
87 residual stands (Pinard and Putz, 1996; Putz et al., 2001; but see Sist et al., 2003). If this is true, RIL
88 may be able to achieve similar timber yields to conventional methods whilst reducing losses of tree
89 biodiversity and the amount of stored carbon.

90 While recent meta-analyses of the impacts of logging in tropical forests have suggested that
91 differences in methods and logging intensity are important for explaining observed differences in
92 responses of animal populations (Bicknell et al., 2014) and species richness (Burivalova et al.,
93 2014), there has been no such meta-analysis on the impacts of logging on trees this. This, despite
94 such evidence being important for informing management of logged tropical forests, especially for
95 programmes such as REDD+ that focus on carbon storage and sequestration. In this study, we aim
96 to address this knowledge gap by conducting a meta-analysis to determine what factors relating to
97 logging method and intensity might explain differences in (1) residual stand damage, (2)
98 aboveground biomass loss, and (3) tree species richness.

99

100 **Methods**

101

102 **Systematic review**

103

104 We defined selectively logged tropical forests as native forests between the latitudes of 40°N
105 and 40°S (Newbold et al., 2013) that have been subjected to the selective removal of trees for
106 timber. We undertook a systematic review using standard methods (Pullin and Stewart, 2006) as
107 outlined in the supplementary materials. Once the search was completed, irrelevant papers were
108 discarded. Our inclusion criteria were that: (i) studies must present data on residual stand damage
109 following logging or above-ground tree biomass and/or species richness of trees from at least one

110 undisturbed forest and one logged forest site; (ii) sites should have spatially replicated measures of
111 the metrics of interest in both logged and unlogged sites with at least three plots present in each.
112 This rule was relaxed for the studies of residual stand damage since very few were replicated or had
113 comparisons with unlogged sites; (iii) logged sites could not be affected by multiple disturbance
114 types, such as fire; and (iv) studies were carried out in terrestrial forests, excluding mangroves.

115 For articles that measured changes in biomass or species richness we extracted the mean,
116 standard deviation and sample size of metrics in both logged and unlogged forests. For studies of
117 forest damage we extracted the mean of each metric used to assess damage. We also recorded the
118 geographic location (latitude and longitude), region (Americas, Africa, or Asia), method of logging
119 used (RIL or conventional selective), the number of years since logging, and volume of wood
120 extracted (m^3ha^{-1}) and/or number of trees felled per hectare. For sites that had been logged twice,
121 we calculated logging intensity as the sum of the volume extracted over both cycles, following
122 Edwards et al (2013). For details of studies used see the supplementary materials.

123

124 **Statistical Analysis**

125

126 **Calculation of metrics of damage and intensity**

127 Prior to analysis we standardised metrics of stand damage and logging intensity. Metrics
128 used to measure residual stand damage fall into two groups: tree-based measurements and area-
129 based measurements (Picard et al., 2012). While tree-based measures attempt to identify the number
130 or proportion of trees damaged per hectare or per tree felled, area based measurements aim to
131 identify the area or proportion of total area of plots in which trees have been damaged. Conversion
132 between the two types of metric is difficult since they show non-linear relationships (Picard et al.,
133 2012), so for this study we concentrated on studies that directly measured damage of trees.

134 We used the proportion of residual trees damaged after logging as our metric of stand
135 damage. Following Picard et al. (2012) we used linear mixed models to determine the relationship
136 between the number of trees damaged per hectare and the proportion of residual trees damaged
137 where both had been measured. The continent on which studies were undertaken was included as an
138 interaction in these models since forest stem density and tree size varies considerably across the
139 tropics (Slik et al., 2013), and therefore the slopes of these relationships could be expected to vary
140 by continent. This model was then used to predict the proportion of residual trees damaged in
141 studies where such data were not directly available. A similar process was undertaken to convert
142 metrics of logging intensity to the metric used in this study, m^3 wood removed ha^{-1} . The number of
143 trees harvested per hectare was the only other metric commonly used and therefore a linear mixed
144 model of the relationship between this variable and the volume of wood removed was produced,

145 accounting for continent level differences in this relationship. Where data on volume of wood
146 removed were not available from a study values were imputed using coefficients from this model.
147 All of these models used a Gaussian error distribution.

148

149 **Impacts of logging on damage, biomass loss and species richness**

150 To determine the effect of logging intensity and different logging methods on the proportion
151 of residual trees damaged, an unweighted linear mixed model was used. Prior to model fitting the
152 response variable was logit transformed so that values were strictly constrained between 0 and 1
153 (Warton and Hui, 2011). Random effects were used to identify data from the same study since their
154 responses are likely to be more similar to each other than those of forests from different studies. We
155 tested how logging volume affected the proportion of residual trees damaged, and whether logging
156 method changed the slope of this relationship. Previous work by Picard et al. (2012) suggested that
157 the relationship between logging damage and intensity is non-linear and so models with quadric and
158 log terms were also tested. R^2 statistics were obtained using the method of Nakagawa and
159 Schielzeth (2013).

160 For the analysis of the impact of logging intensity and logging method on changes in above-
161 ground biomass and species richness, a weighted approach was used. Where standard errors of the
162 mean were missing from studies, they were estimated using imputation methods (Koricheva et al.,
163 2013) which are likely to bias results less than excluding studies with incomplete information
164 (Nakagawa and Freckleton, 2008). To achieve this, the relationship between the coefficient of
165 variation for logged and unlogged sites for tree richness or biomass and plot size at which data were
166 collected was estimated using linear models. The literature on human-disturbed forests suggests that
167 smaller sampling plots result in greater between-sample variation and therefore higher coefficients
168 of variation, indicating that this approach is empirically supported (Wagner et al., 2010).
169 Unweighted linear models were then used to predict the coefficient of variation for studies missing
170 these data, and missing standard deviations were calculated by multiplying this prediction by the
171 value of richness or biomass measured at the site.

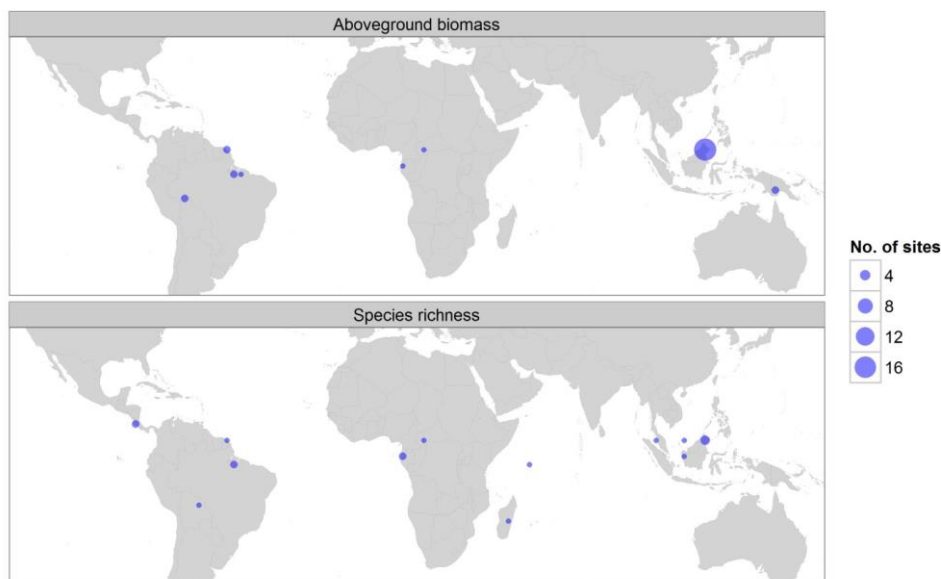
172 To analyse the effects of logging on carbon pools and tree species richness, the log response
173 ratio of differences between sites was calculated and models were weighted so that more precise
174 studies had more weight (Borenstein et al., 2009; Hedges et al., 1999). We fitted a random effects
175 meta-regression to account for pseudo-replication at the level of individual studies when the same
176 unlogged site was used as a comparator for multiple logged sites. In the analyses of richness,
177 estimation method (rarefied or not rarefied) was included as a random effect since this has been
178 shown to cause between-study differences in the past (Cannon et al., 1998; Gotelli and Colwell,
179 2001), but the nature of any difference was not a focus for this study.

180 We tested the effects of logging method and logging intensity in determining post-logging
181 biomass and changes in tree species richness. It is also possible that the time since a site was last
182 logged and the location of study may play a role in determining logging impacts (Burivalova et al.,
183 2014) and so these variables were also included in models. All plausible models that had >3 data
184 points per parameter were assessed and R^2 values were calculated (see supplementary materials for
185 details of all models tested).

186 In model selection AICc was used to determine the relative likelihood of a model being the
187 'best model'. All models of tree damage with a $\Delta AICc < 7$ were averaged to produce coefficient
188 estimates (Burnham et al., 2011). Since this is not possible for the weighted analyses of impact on
189 biomass and species richness, the model with the lowest AICc was chosen as the most parsimonious
190 model. All statistical analyses were performed in R 3.0.2 (R Development Core Team, 2011), with
191 unweighted analyses carried out using the lme4 package (Bates et al., 2014), weighted analysis
192 carried out using the package metafor (Viechtbauer, 2010) and figures drawn using ggplot2
193 (Wickham, 2009).

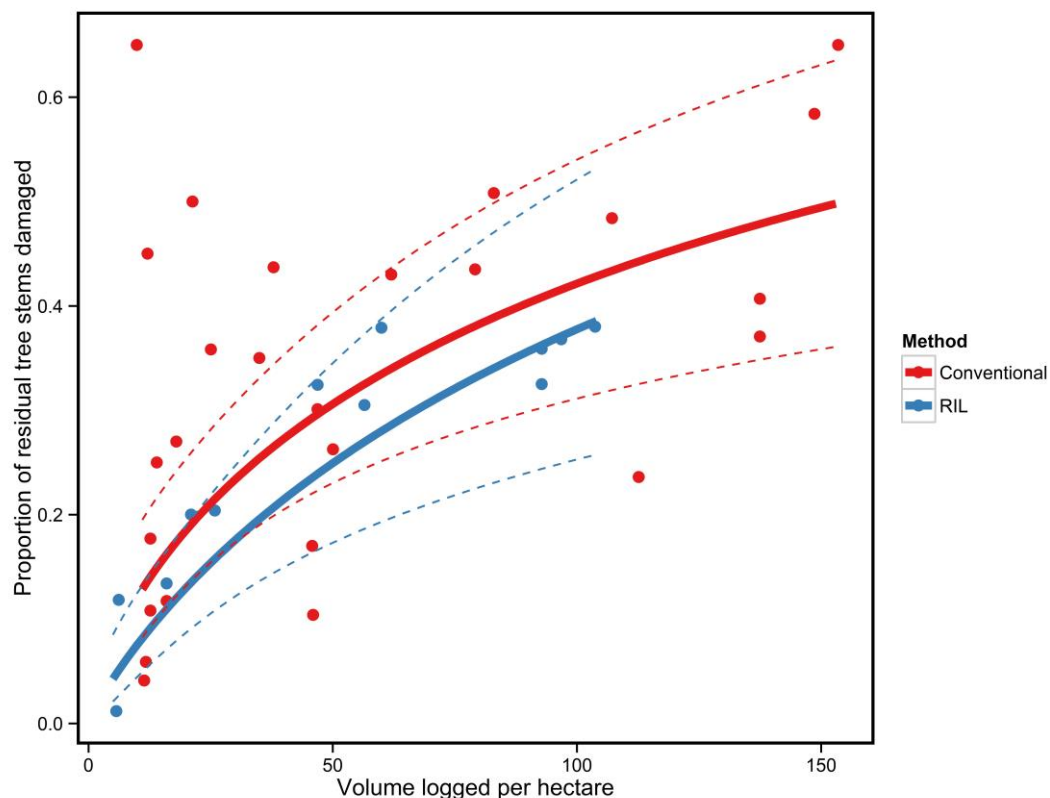
195 Results

196 The systematic review yielded 62 studies, from which we extracted data on residual tree
197 damage from 38 sites, and 43 and 9 paired, replicated sites that measured biomass and tree species
198 richness respectively. Median logged-site age for those sites where biomass was measured was 4.5
199 years (min=0, max=30) and for sites where richness was measured it was 5 years (min=0, max=50).
200 Sites were mostly located in Asia and South America, with relatively few in Africa (Figure 1).



201
202 Figure 1 - Geographic location of studies used in our analyses on change in aboveground biomass
203 and species richness in response to logging.

204 The most parsimonious model for predicting the volume of wood logged per hectare
 205 comprised as predictors the number of trees extracted per hectare and the continent where
 206 studies were undertaken, with an interaction between the two and had a high explanatory power
 207 ($R^2=0.93$). This model indicated that for each tree removed a greater volume of wood was removed,
 208 the slope of the relationship being highest in Asia followed by Africa and the Americas (Figure S1).
 209 The most parsimonious model for converting from the number of trees damaged per hectare to the
 210 proportion of trees damaged per hectare did not require inclusion of study location as a variable and
 211 also had high explanatory power ($R^2=0.95$, Figure S2).
 212



213
 214 Figure 2 - Impact of selective logging intensity and logging technique on the proportion of residual
 215 tree stems damaged in tropical forests (n=38). Points represent single sites, solid lines are the
 216 predictions the most parsimonious linear mixed effects model ($R^2=0.45$) and dashed lines represent
 217 the 95% confidence intervals of these estimates. Red points and lines refer to sites where
 218 conventional harvest methods were used and blue points and lines where RIL techniques were used.
 219 For details of alternative models considered see Table S1.

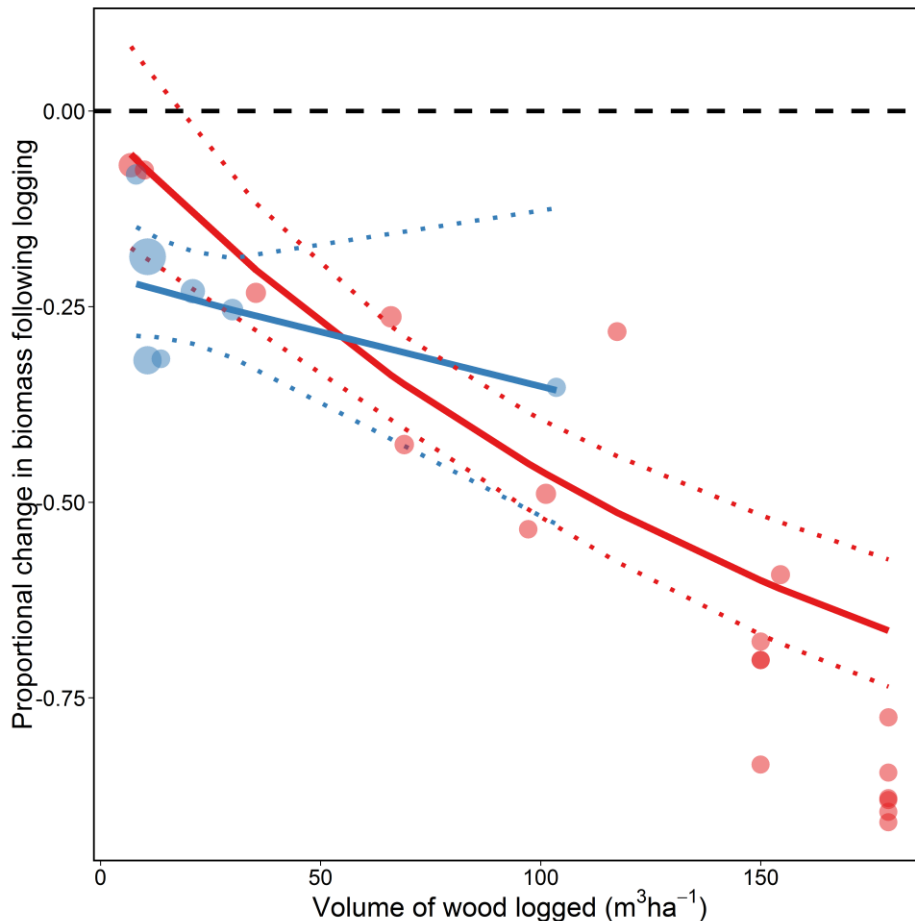
220

221 Residual stand damage

222 The model that best explained the proportion of residual tree stems that were damaged
 223 included an interaction between the logarithm of logging intensity and the logging method

224 ($R^2=0.45$). No other models had a $\Delta AICc < 7$ (Table S1). Predictions suggested that damage to the

225 residual logging stand increased as a function of the logarithm of the logging intensity (Figure 2).
226 This model also suggested that at low logging intensities, RIL tended to cause less residual damage
227 than conventional logging, but at high intensities the two methods became more similar in the
228 residual damage that they caused (Figure 2). However, the 95% confidence intervals for predictions
229 were very wide indicating large variation in damage on residual tree stems for both methods.



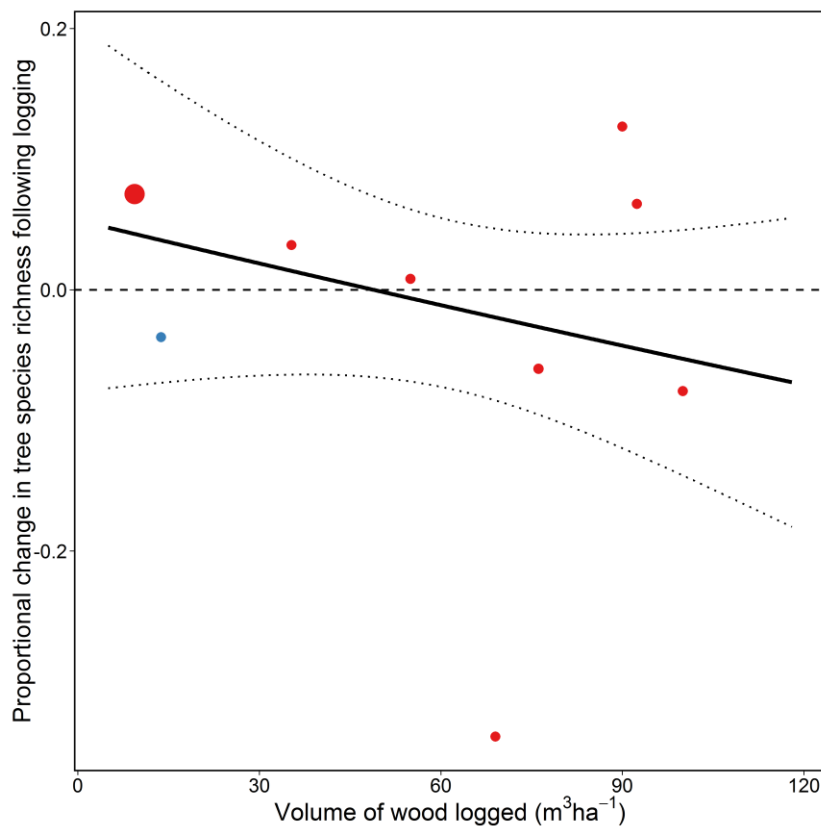
230
231 Figure 3 – Impacts of selective logging intensity on changes in aboveground biomass (n=32). Blue
232 symbols are those sites where reduced impact logging (RIL) was carried out, red symbols
233 correspond to conventionally logged sites with size indicative of study weight. The solid lines
234 represents the predictions from the weighted meta-regression model with lowest AICc ($R^2=0.96$),
235 and the dotted lines the corresponding 95% confidence intervals. The horizontal dashed line
236 indicates when there is no difference between logged and unlogged sites. Note that RIL sites tend to
237 be logged at a lower intensity than conventionally logged sites. For details of alternative models
238 considered see Table S2.

239 240 **Impacts of logging intensity and method on biomass and species richness**

241 Increased logging intensity led to reduced post-logging biomass and the slope of this
242 relationship differed between RIL and conventional methods (Figure 3, Table S2). The model

243 explaining variation in biomass effect size with greatest support ($R^2=0.96$, Table S2) suggested a

244 linear relationship between logging intensity and changes in biomass and an interaction between
245 this and logging method. The model indicated that RIL techniques may result in lower biomass
246 losses per m^3 of wood removed per hectare at intensities $>50m^3 ha^{-1}$, but higher biomass losses
247 below this (Figure 3). However the relatively low intensities at which RIL sites tended to be logged
248 compared to conventional sites reduced statistical power, and confidence intervals for the
249 predictions for RIL and conventional logging overlapped.



250

251 Figure 4 - Impacts of logging intensity on tree species richness ($n=9$) following logging. Blue
252 symbols are those sites where RIL was carried out, red symbols correspond to conventionally
253 logged sites with point size indicative of study weight. The solid line represents the predictions
254 from the weighted meta-regression model with lowest AICc ($R^2=0.36$), and the dotted lines the
255 corresponding 95% confidence intervals. The horizontal dashed line indicates when there is no
256 difference between logged and unlogged sites. For details of alternative models considered see
257 Table S3.

258

259 The model that explained variation in tree species richness effect size most effectively
260 suggested a negative relationship with intensity of logging, with a positive intercept (Figure 4, Table
261 S3). Only one other model had a $\Delta AICc < 7$ that included intensity and intensity squared, and the
262 most parsimonious model had an R^2 of 0.36. Too few studies assessed the impact of RIL on species
263 richness to conduct an analysis of its effect relative to conventional logging.

264 **Discussion**

265

266 Our results indicate that the impacts of selective logging in tropical forests on residual stand
267 damage, biomass loss and species richness change are largely explained by differences in logging
268 intensity. Residual tree damage also appears to be reduced at lower intensities under RIL when
269 compared to conventional logging. However the effect of RIL on biomass loss was difficult to
270 assess owing to the confounding effects of differences in logging intensity.

271

272

273 **Impacts of logging on stand damage and biomass**

274 Our meta-analysis indicates that logging intensity is the primary driver of differences in
275 non-target tree damage in selectively logged tropical forests, as previously noted in other studies
276 (Picard et al., 2012; Sist et al., 1998). However, our results from a wide range of sites also support
277 the hypothesis that RIL causes lower damage to residual trees than conventional logging,
278 especially at lower intensities. Similar observations have been made in the field by Sist et al. (1998)
279 who indicated that RIL reduced residual damage by around 50% when logging was carried out at
280 intensities $<65 \text{ m}^3 \text{ ha}^{-1}$ in Indonesian Borneo, but that values were similar to conventional logging at
281 higher intensities.

282 Given that residual damage to trees is likely to account for the majority of carbon losses as a
283 result of selective logging, this indicates that RIL may be useful in reducing carbon losses at the
284 stand scale (Gourlet-Fleury et al., 2013b; Mazzei et al., 2010). However, there is a large amount of
285 between-study variation in the impacts of logging intensity and methods on stem damage, which
286 suggests either that other variables that are important, such as the density of log extraction routes or
287 the steepness of slopes where logging was undertaken (Putz et al., 2000), have not been considered
288 in our models. Equally, this variation may be a result of the large variation in between-study
289 methods used to assess stem damage.

290

291 **Impacts of logging on aboveground biomass**

292 Our analyses emphasize that accounting for harvesting intensity is vital for understanding
293 effects of logging on above-ground biomass. The volume of wood removed per hectare was by far
294 the best predictor of changes in biomass in response to timber harvest. Logging intensity varies by
295 region (Figure S3 & Putz et al., 2001), and is relatively high in Asia (mean $99.3 \text{ m}^3 \text{ ha}^{-1}$ in this
296 study) compared to South America and Africa (mean 31.3 and $17.9 \text{ m}^3 \text{ ha}^{-1}$ respectively). This
297 variation in intensity may reflect the fact that SE Asian forests are largely dominated by dipterocarp
298 trees, which have high timber value (Corlett and Primack, 2005), and thus contain a larger number

299 of harvestable timber trees per hectare compared to other regions. The combination of higher
300 logging intensities and higher above-ground biomass in SE Asian forests than in other areas of the
301 tropics (Slik et al., 2013) is likely to result in greater per ha carbon emissions from forests logged in
302 SE Asia than Africa or Latin America.

303 We find weak support for the hypothesis for a difference between the impacts of RIL and
304 conventional selective logging on post-logging biomass. Although models suggested a difference in
305 the relationship between logging intensity and biomass loss for RIL and conventional logging this is
306 driven by relatively few data points (Figure 3, n=7 for RIL). This lack of data from studies of RIL
307 and the relatively low logging intensities at which RIL is carried out when compared to
308 conventional selective logging make firm conclusions about this relationship difficult (Figure S3).
309 Unless studies of RIL are carried out at a similar range of intensities to conventional selective
310 logging its potential carbon benefits, aside from those resulting from lower logging intensities, are
311 almost impossible to assess.

312

313 **Impacts of logging on species richness**

314 As for above-ground biomass, logging intensity best explained differences in tree species
315 richness caused by logging. However, compared to above-ground biomass, the slope of this
316 relationship was much less steep, with an apparent initial increase in species richness at low
317 intensities. The most plausible explanation for this increase is an influx of generalist species from
318 surrounding non-forest areas (Carreño-Rocabado et al., 2012) leading to an initial post-harvest
319 increase in richness. Similar relationships have recently been observed between logging intensity
320 and bird species richness, while other vertebrates showed a decline even at low intensities
321 (Burivalova et al., 2014).

322 Our results suggest that tree species richness may be relatively insensitive to subtle changes
323 in forest cover (Cannon et al., 1998). However, changes in species richness provide no information
324 about the identity and function of individual species. Community composition is likely to be
325 impacted by selective logging, with forest-dependent species sensitive to disturbance becoming less
326 abundant or locally extinct (Sheil et al., 1999) and generalist species increasing in abundance
327 (Baraloto et al., 2012; Gourlet-Fleury et al., 2013a). However, analysis of logging impacts on
328 community composition is hindered because most studies of logging are spatially pseudoreplicated,
329 leading to biased estimates of change (Ramage et al., 2013).

330

331 **Improving assessments of logging intensity and damage**

332 Our analyses support conclusions by others (Bicknell et al., 2014; Burivalova et al., 2014)
333 that consideration of logging intensity is vital to understand the impact of logging on biodiversity

334 and above-ground biomass. However, it can be difficult to obtain statistics on the volume of wood
335 removed from an area, and when such data are available they are often only available as a mean
336 volume removed per hectare for the entire study area. For individual studies, identification of the
337 importance of logging intensity is extremely difficult. To solve this, the use of metrics of logging
338 intensity such as basal area logged ha^{-1} may prove fruitful. Previous studies have used such metrics
339 to examine the importance of logging intensity in biomass recovery rates (Mazzei et al., 2010). This
340 has the advantage of allowing an estimate of logging intensity at the plot scale, allowing for more
341 nuanced analyses of logging impacts than is currently possible.

342 A wide variety of different measures is used to assess residual logging damage in selectively
343 logged forest stands (Picard et al. 2012), fostered by different objectives and hypotheses. We used
344 coefficients to convert between different measures to maximise the value of available data, but this
345 method inevitably introduced additional uncertainties into our analyses. Synthesis would be aided
346 by standardisation of metrics. We suggest the use of standardised metrics when assessing tree
347 damage and recommend that assessments of damage should be carried out at the level of individual
348 trees rather than assessing the proportion of area affected by logging activities. We also suggest that
349 future studies should report the proportion of basal area that is damaged to provide additional
350 information of logging impacts on forest biomass. Furthermore, stratification of logging damage by
351 tree size class would allow an assessment of its potential demographic effects and would therefore
352 aid our understanding of the recovery of logged forests.

354 **Reducing the negative effects of logging**

355 Our meta-analysis suggests that the most obvious way of reducing the negative impacts of
356 tropical logging is to reduce local logging intensity. These findings are in contrast to a recent
357 extensive meta-analysis, which suggested that RIL at $<30 \text{ m}^3 \text{ ha}^{-1}$ resulted in larger animal
358 populations than conventional logging at similar intensities (Bicknell et al., 2014). The possible
359 reasons for this apparent difference in drivers of logging impact are twofold. Firstly, the work of
360 Bicknell et al. (2014) addressed animal population sizes which are potentially affected to a greater
361 degree by logging at the landscape scale than the metrics we examined for trees. For example forest
362 specialist birds in the Amazon are sensitive roads $<10\text{m}$ in width, resulting in reduced patch
363 occupancy for selected species, while tree biomass and species richness have been seen to recover
364 for relatively isolated secondary forests (Martin et al., 2013), though species composition may differ
365 because of dispersal limitation (Chazdon, 2003). RIL results in much lower road densities than
366 conventional logging, thus reducing gaps that animals need to cross as well as edge effects and
367 forest loss at the landscape scale. Secondly Bicknell et al. (2014) used a higher sample size than
368 was available for our meta-analysis. Inclusion of new studies in a meta-analysis such as ours would

369 help to clarify this observed disparity.

370 The results of our study and those of Burivalova et al. (2014) suggest that logging intensity
371 drives carbon and species loss while Becknell et al. (2014) suggest that RIL is less damaging for
372 animal populations. As such, current evidence suggests that RIL at relatively low intensities is likely
373 to be the best way to reduce carbon and biodiversity loss in tropical logged forests. However, given
374 the massive area of tropical forest already designated for logging (Asner et al., 2009), reductions in
375 local intensity, and thus yield, may encourage expansion into previously unlogged areas. This
376 mirrors the situation in agricultural landscapes where the biodiversity benefits of high-yield farming
377 over small areas as opposed low-yield, extensive farming is debated (Benayas et al., 2012; Phalan et
378 al., 2011). Recent work indicates that high intensity logging over a smaller area ('land sparing') has
379 better outcomes for tropical forest species than low-intensity extensive timber extraction ('land
380 sharing') in Borneo (Edwards et al., 2014). This sparing/sharing framework may prove useful to
381 assess the potential value of differing land-use strategies in landscapes used to provide ecosystem
382 services such as food and timber.

383 Although reductions in logging intensity may reduce impact, the high demand for timber
384 requires novel solutions that do not drastically reduce current yields but reduce impacts on forest
385 ecosystems. Methods such as silvicultural thinning techniques to remove pioneer species may aid
386 recovery of floral community composition, carbon and timber stocks but further work is needed to
387 assess their effectiveness (Gourlet-Fleury et al., 2013b; Ouédraogo et al., 2011). Although RIL may
388 also provide a solution, further evidence is required to verify this for carbon storage in the form of
389 above-ground biomass. Analyses that take into account plot level variation in logging intensities
390 using collaborative networks such as The Tropical managed Forests Observatory (Sist et al., 2015)
391 offer a potential solution to this.

392

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398

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