

Agroforestry boosts soil-mediated ecosystem services in the humid and sub-humid tropics: a meta-analysis

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1 **Agroforestry boosts soil-mediated ecosystem services in the humid and sub-humid tropics: a**
2 **meta-analysis**

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19

20 ABSTRACT

21 Agroforestry has been increasingly recognized as a key example of agroecological praxis contributing to
22 the sustainable intensification of food production while providing a number of additional benefits to
23 society. However, a quantitative synthesis of the impact of agroforestry on soil health and associated
24 ecosystem services in the humid and sub-humid tropics is still lacking. Therefore, the objective of this
25 study was to quantify the contribution of agroforestry practices to soil-mediated ecosystem services,
26 specifically, regulation of soil erosion, storage of soil organic carbon (SOC) and nitrogen (N), availability
27 of soil N and phosphorus (P) to crops, and alleviation of soil acidity across the humid and sub-humid
28 tropics. The analysis demonstrated that agroforestry can reduce soil erosion rates by 50% compared to
29 crop monocultures. This finding is supported by higher infiltration rates, lower runoff, higher proportion
30 of soil macroaggregates, and greater stability of soil structure under agroforestry. SOC increased by 40%,
31 N storage increased by 13%, available N by 46% and available P by 11% while soil pH increased by 2%
32 under agroforestry compared to crop monocultures. We conclude that agroforestry can make significant
33 contributions to provision of soil-mediated ecosystem services in the humid and sub-humid tropics.

34

35 **Key words:** agroecology; indicators; soil health; tropical agriculture

36

37

38 **1. Introduction**

39 Agricultural intensification has been responsible for net gains in human well-being and economic
40 development, but with an increasing cost of degradation of natural resources (Matson et al., 1997; MA
41 2005). This realization has led to a growing demand for agroecological approaches that support
42 intensification trajectories which can be sustained in the long term to feed an estimated global population
43 of 9.7 billion people by 2050 and 11.2 billion people by 2100 (Lal, 2016; O’Neill et al., 2018). Many of
44 the ecological intensification approaches increase soil organic matter (SOM) and soil-based ecosystem
45 services enhancing sustainability of agricultural systems (Barrios, 2007; Garratt et al., 2018). In this
46 context, agroforestry, which embraces the multiple forms of interactions between trees and crops, has
47 been increasingly recognized as a promising intensification pathway aiming at sustainable agriculture
48 (Pretty 2018). Agroforestry has been described as agroecology in practice because it successfully adapts
49 ecological concepts and principles to the design and management of agroecosystems (Gliessman, 2007;
50 Prabhu et al., 2015). Agroecology has received recent recognition as a holistic approach centrally
51 contributing to the achievement of the Sustainable Development Goal 2 (SDG 2) targets of ending
52 hunger, achieving food security and improved nutrition, and promoting sustainable agriculture (FAO,
53 2017). Agroforestry as a land management option can simultaneously contribute to household income,
54 food security and the conservation of biodiversity and ecosystem services (Akinnifesi et al., 2010; Fonte
55 et al., 2010; Kamau et al., 2017; Barrios et al., 2018). It can also serve as a climate change mitigation and
56 adaptation tool for agriculture (Mutuo et al., 2005; Verchot et al., 2007; Schoeneberger et al., 2012).

57 Two major types of agroforestry practices can be distinguished: i) simultaneous agroforestry where
58 trees and crops occur on the same piece of land during the same cropping season (e.g. shaded coffee and
59 cocoa systems, alley cropping, intercropping), and ii) sequential agroforestry where trees and crops occur
60 on the same piece of land but in a temporal sequence as part of a rotation (e.g. improved fallows) (Sanchez

61 et al., 1997; Sinclair, 1999). These agroforestry practices are expected to have widely differing impacts
62 on soil-based ecosystem services (Figure 1).

63

64 (Insert Fig. 1)

65

66 The temporal and spatial arrangement of trees are likely to have different impacts on soil health
67 indicators and soil-based ecosystem services. While ‘soil quality’ and ‘soil health’ have often been used
68 interchangeably, we recognize here that they reflect a shift in conceptual thinking from a focus on soil
69 physical and chemical properties towards an increasing recognition of the soil as a living entity in which
70 soil biological properties play a critical role in the adaptation to global change (Barrios et al., 2015). Soil
71 health is defined here as “an integrative property that reflects the capacity of soil to respond to agricultural
72 intervention, so that it continues to support both the agricultural production and the provision of other
73 ecosystem services” (Kibblewhite et al., 2008). Soil health is one of the three components of
74 environmental quality besides water and air quality (Doran, 2001; FAO and ITPS, 2015).

75 Soils provide many ecosystem services in all the three main categories, namely provisioning
76 services, regulating and maintenance services and cultural services (Palmer et al., 2017; Robinson et al.,
77 2014). Through enabling plant growth, soils provide human food, animal feed, fiber, energy and genetic
78 materials. The regulation and maintenance services provided by soils include nutrient storage and supply,
79 sequestration of greenhouse gases, flood mitigation, biological control of pests and diseases, adsorbing
80 and detoxifying harmful chemicals. Soils also provide cultural services, which include non-material and
81 non-consumptive benefits that affect the physical and mental state of people. At the 23rd Conference of
82 the Parties to the UNFCCC held in November 2017, countries recognized the fundamental importance
83 of soil carbon, soil health and soil fertility in responding to climate change with the dedicated Koronivia
84 Joint Work on Agriculture (FAO, 2018).

85 The focus of this meta-analysis is on the humid and sub-humid tropics. This focus was motivated
86 by several factors, but the main ones are (1) the greater potential for productivity increase to meet future
87 food demands in this regions than in other parts of the world; (2) a large proportion of the rural population
88 in the humid and sub-humid tropics faces significant soil degradation (Barret and Bevis, 2015), and (3)
89 this region faces the greatest risks to global biodiversity losses (Myers et al., 2000) partly driven by
90 agricultural expansion into forest land and common practices such as shifting cultivation (Heinimann et
91 al. 2017). Humid and sub-humid tropical regions are also dominated by low-activity clay soils which
92 suffer from soil acidity and associated toxicities, low nutrient reserves and multiple nutrient deficiencies,
93 and are prone to erosion particularly on exposed sloping land (IUSS, 2014). The potential for agroforestry
94 to alleviate many of these constraints and increase food production, improve human nutrition and health,
95 and conserve of natural resources is higher in the humid/sub-humid tropics than elsewhere (Nair and
96 Garrity, 2012).

97 Reviews and meta-analyses published in the last three decades have increased our understanding
98 of the impact of agroforestry on some of the provisioning services such as crop yields (e.g. Bayala et al.,
99 2012; Kuyah et al., 2016; Sileshi et al., 2008), and regulating services such as control of pests, diseases
100 and weeds (e.g. Pumariño et al., 2015) and carbon sequestration (e.g. Chatterjee et al., 2018, Cerda et al.
101 2019). However, similar reviews and meta-analysis do not exist on the mechanisms by which
102 agroforestry practices impact on soil health and soil-mediated ecosystem services. This synthesis was
103 designed to address research questions and hypotheses (Table 1) that have remained outstanding and
104 were not addressed by earlier syntheses and meta-analyses. Although several studies assessing the effects
105 of agroforestry on various soil properties have been published in the last three decades, a quantitative
106 synthesis of the results from those studies is still lacking. Therefore, the objective of this meta-analysis
107 is to quantify the contribution of agroforestry practices to soil-mediated ecosystem services, specifically,
108 regulation of soil erosion, storage of soil carbon (C), storage of soil nitrogen (N), availability of soil N,

109 availability of soil phosphorus (P), and alleviation of soil acidity across the humid and sub-humid tropics.
110 The overall aim of this synthesis is to create awareness among researchers, development practitioners
111 and policy-makers on the roles that agroforestry can play in climate change adaptation and mitigation as
112 well as management of land degradation. This kind of information is hoped to be useful as countries
113 engage in the Koronivia Joint Work on Agriculture and the preparation of their next nationally
114 determined contributions (NDCs) to the UNFCCC.

115 116 **2. Methods**

117 *2.1. Selection of indicators*

118 To facilitate the analyses, first we identified key indicators of the ecosystem services mentioned
119 above. The term “indicator” is frequently used at the interface between science and policy (Heink and
120 Kowarik, 2010), and indicators are often used to describe, represent, monitor, assess or model complex
121 processes or system properties to be used in decision-making. No consensus exists on practical indicators
122 for soil ecosystem services (Rutgers et al., 2012). Several chemical, physical and biological variables
123 may be used as indicators for ecosystem services (Rutgers et al., 2012). We chose a set of indicators
124 based on (1) their high frequency of reporting in published studies to enable the use of meta-analytical
125 tools and (2) their ability to represent major soil health constraints globally (FAO and ITPS, 2015). A
126 recent review by Barrios et al. (2012) highlighted that the limited number of studies conducted on the
127 impacts of agroforestry on soil biological parameters still limits their use in meta-analysis, hence this
128 quantitative synthesis focussed on key physical and chemical indicators. Specifically, we focused on
129 indicators of soil erosion rate, namely eroded soil, infiltration rate, run off, macroagregates and mean
130 weight diameter (MWD). Aggregate stability is a measure of how well soil aggregates resist
131 disintegration when hit by rain drops, and it is a key indicator of resistance to erosion. For soil C storage
132 we limited the analysis to soil organic carbon (SOC) and macroaggregate-associated C. The critical
133 importance of SOC to support the provision of multiple ecosystem functions and services has been widely

134 acknowledged (FAO and ITPS, 2015). Indeed, SOC is considered a key indicator of soil health, a
135 universal proxy of multiple ecosystem services and an important driver of agricultural sustainability (Lal,
136 2015; Palmer et al., 2017; Rutgers et al., 2012). In the case of soil N storage, we focussed the analysis
137 on total N and macroaggregate-associated N. Total N allows assessment of the contribution of
138 agroforestry to soil N stocks (Johnson and Curtis, 2001).

139 The availability of nutrients that limit productivity is another important indicator of regulating and
140 supporting ecosystem services. Here we used ammonium-N, nitrate-N and available N (ammonium-N +
141 nitrate-N) as the key indicators of soil N availability, and available P as the key indicator of soil P
142 availability. In the case of amelioration of soil acidity, we used soil pH as the key indicator. This is
143 because soil pH has a direct influence on physical, chemical (e.g. nutrient availability, toxicity) and
144 biological (e.g. microbial activity) characteristics that influence crop growth.

145

146 *2.2. Research questions and hypotheses*

147 Building on earlier reviews and syntheses (e.g. Sanchez et al., 1997; Buresh and Tian, 1998; Van
148 Noordwijk et al., 2004; Barrios et al. 2012), five research questions and associated hypotheses were
149 developed to guide this meta-analysis. The research questions were: (1) Does agroforestry reduce soil
150 erosion? (2) Does agroforestry build soil organic C and N stocks? (3) Does agroforestry increase soil N
151 availability? (4) Does agroforestry increase soil P availability? and (5) Does agroforestry alleviate soil
152 acidity? Under each question, we tested several hypotheses (Table 1), some of which have been proposed
153 by other researchers, but have remained untested.

154

155 (insert Table 1)

156

157 *2.3. Literature search*

158 The meta-analyses aimed at comparing soil properties associated with sequential or simultaneous
159 agroforestry practices with those associated with the corresponding crop monocultures. Therefore, our
160 literature search focussed on studies that compared plots where crops were associated with trees
161 (agroforestry) with plots where crops were grown without trees (crop monocultures). Publications for the
162 meta-analysis were first identified using the ISI Web of Science focusing on literature published up to
163 July 2017. We searched published studies that reported the effects of agroforestry on soil health covering
164 the aggregate ecosystem functions of C-transformations, soil structure maintenance and nutrient cycling
165 (Barrios et al., 2012). Two searches were conducted using 20 keywords on different agroforestry
166 practices in combination with either 19 or 25 key words representing response variables associated with
167 soil structure maintenance/soil C storage and nutrient cycling, respectively (Supplementary Table S1).
168 This was followed by an intensive review of abstracts and papers to be included in the meta-analysis. A
169 total of 119 articles qualified for the meta-analysis (Supplementary Table S2). We also examined the
170 reference lists of papers including previous syntheses on related topics. As part of our data compilation
171 the following factors were included in the database: location (country, latitude, longitude and altitude),
172 mean annual rainfall, soil type (WRB classification), soil texture, agroforestry practice (i.e. simultaneous
173 or sequential), tree species, crop species, study type (experimental or observational), soil response
174 variable (e.g. soil available P), soil depth, data collected in both control and intervention treatments
175 (mean, SE, SD, n).

176

177 *2.4. Criteria for inclusion in meta-analysis*

178 For a study to be included in this meta-analysis, it had to fulfil the following criteria:

- 179 1. The study originated in the humid or sub-humid tropics (annual rainfall >600 mm, within 30°
180 North/South of Equator).

181 2. The study compared plots representing one or more simultaneous or sequential agroforestry
182 practices with plots of crop monocultures (the monoculture plots will henceforth be referred to as
183 “control”). Agroforestry and control plots were located on the same farms and the only difference in
184 farming practice between the two plots was the presence or absence of trees. Agroforestry practices were
185 classified into simultaneous and sequential practices. Studies in which the agroforestry practice involved
186 organic inputs coming from outside (e.g. biomass transfer systems) or in which the tree effect could be
187 confounded with other inputs (e.g. manure inputs as in silvopastoral systems) were excluded from the
188 analysis. Furthermore, rotational woodlots (trees grown >3 years) and home-gardens, often classified as
189 agroforestry practices, were excluded from the current analysis due to lack of studies reporting a proper
190 control plot.

191 3. The study had the same crop species grown in the agroforestry plot and the corresponding control
192 plot.

193 4. The study quantified one or several of the indicators of aggregate ecosystem function and soil
194 health highlighted in section 2.1.

195 5. Only studies conducted at the farm scale were included, hence those at landscape scale and in
196 the laboratory were excluded.

197

198 *2.5. Data extraction*

199 From each publication that qualified for the meta-analysis, we extracted data on soil erosion,
200 infiltration, runoff, % macroaggregates, MWD, soil organic carbon (SOC), total N, macroaggregate C,
201 macroaggregate N, soil available N, nitrate-N, ammonium-N, soil available P and soil pH. Whenever
202 reported individually, soil ammonium-N and/or nitrate-N were discriminated from soil available N which
203 in the literature represents the sum of the two (i.e. ammonium-N + nitrate-N). Only available P data
204 extracted by the Olsen, Bray or Mehlich methods were included in the meta-analysis. The loosely bound

205 P (resin P or water-soluble P) and tightly bound P (P extracted by HCl or sulphuric acid) were not
206 included. Total P was rarely reported in selected articles, hence was not considered.

207 In addition to the data on variables reflecting soil quality and functioning, other ancillary data
208 including geographic coordinates, altitude, mean annual precipitation, soil type and texture were
209 extracted. Soil texture categories were based on the texture triangle (Shirazi and Boersma, 1984) and
210 consider sandy soils (< 20% clay), loam soils (20-32% clay), clay soils (> 32% clay).

211 Data were extracted from the results section, tables, appendices, graphs and figures from each of
212 the papers. Data from graphs were extracted using IMAGE J software. Whenever multiple agroforestry
213 treatments with different tree species were presented in a given paper, each treatment by control
214 comparison was considered as a separate data point in the meta-analysis. We also considered treatments
215 based on different tree species compared with the same control as unique observations (Tonnito et al
216 2006). If a paper reported results from more than one soil depth, only the upper soil layer (till layer) was
217 considered. In cases where tests were repeated over the growth period, we selected the soil measurements
218 made before the last growing season of the experiment to capture the cumulative effects.

219

220 *2.6. Effect size*

221 For all data analyses the response ratio (RR) was used as the effect size. RR is defined as the ratio
222 of the treatment value (T) to the corresponding control value (C) for any given variable, i.e. $RR = T/C$.
223 To satisfy the assumptions of normality and homogeneity of the error variance, we used the logarithm of
224 RR (logRR) for the meta-analysis as recommended by Hedges et al., (1999).

225

226 *2.7. Data analysis*

227 We applied a linear mixed modelling procedure for all analyses. We preferred the mixed modelling
228 approach because many of the studies did not report either the SD or SE. The mixed modelling procedure

229 was also more appropriate as the data gathered across studies were unbalanced with respect to predictor
230 variables and sample sizes. In the mixed model we entered the categorical variables (e.g. agroforestry
231 type, ability to fix N and soil texture) as fixed effects and the source of data (i.e. study) as the random
232 effect. Then we estimated model parameters and their 95% confidence intervals (95% CI) using restricted
233 maximum likelihood estimation (REML). Where moderator variables were not applicable, for example
234 the overall effect of agroforestry on a given variable, the 95% CIs were estimated by bootstrapping
235 (resampling with replacement) with 9999 random replicates.

236 In all cases, the population marginal means and 95% CI of the back-transformed RR are presented.
237 We considered means to be significantly different from one another only if their 95% CI were non-
238 overlapping. Where sample sizes were small (<30), we interpret the results cautiously because the 95%
239 CLs will be wide and prone to Type I error. The 95% CI quantifies both the magnitude and direction of
240 change under agroforestry with respect to the control. If there is no significant difference between
241 agroforestry and the control for a given variable, the 95% CI of RR will encompass 1. On the other hand,
242 if the 95% CL of RR is greater than 1 it means significant increases under the given agroforestry practice
243 over the control. The agroforestry effect was interpreted as significantly negative (leading to reduction)
244 when the 95% CL <1.0. Data on macroaggregate-associated C and macroaggregate-associated N,
245 infiltration rates, runoff and porosity were scarce and, therefore, contrasts of agroforestry management,
246 ability to fix N and soil texture could not been done.

247

248 **3. Results**

249 *3.1. Regulation of soil erosion*

250 Erosion rate were reported in a total of 17 studies and a sample size of 69 was available for analysis.
251 The estimated effect sizes in each study and the overall (all studies combined) are presented in Figures
252 2a. In all studies, RR was less than 1 indicating that soil erosion rates were significantly lower under

253 agroforestry compared to the corresponding crop monocultures. Overall, agroforestry trees reduced soil
254 erosion by 50% (Figure 2a). All the studies on soil erosion were conducted in simultaneous agroforestry,
255 and only one study was found on sequential systems. Although the differences were not statistically
256 significant (Figure 2b), tree species that do not fix N generally contributed to lower erosion (RR = 0.29)
257 than N-fixing trees (RR = 0.41). The effect sizes for erosion rates also did not significantly differ among
258 soil texture classes, but loamy soils had generally lower erosion rates than sandy soils (Figure 2b).

259

260 (Insert Fig. 2a, 2b)

261

262 Infiltration rates were 75% higher under agroforestry than crop monocultures (Figure 3a). Runoff
263 was 57% lower under agroforestry than crop monoculture (Figure 3a). Soil macroaggregates (> 0.25 mm)
264 and mean weight diameter (MWD) were significantly higher under agroforestry than in the crop
265 monocultures (Figure 3b); the increases being 22 and 30% for macroaggregates and MWD, respectively
266 (Figure 3b).

267

268 (Insert Fig. 3a, 3b)

269

270 3.2. Storage of soil carbon

271 SOC was reported in 71 studies and a total of 225 pairs of observations were available for analysis.
272 The estimated effect sizes for each study and the overall mean RR are presented in Figure 4a. With
273 overall effect size of 1.21 (CL: 1.15-1.27), agroforestry significantly increased SOC storage compared
274 to crop monocultures although effects varied with study (Figure 4a). However, the effect size did not
275 significantly differ between simultaneous and sequential agroforestry practices or between N-fixing trees

276 species and those that do not fix N (Figure 4b). SOC storage under agroforestry was significantly greater
277 in sandy soils compared to loamy soils (Figure 4b). Aggregate-associated C was significantly higher
278 under agroforestry than in the crop monocultures (Figure 3b). Closer examination using soil physical
279 fractionation techniques shows that 13-29% greater soil C is stored in macroaggregates under
280 agroforestry practices.

281

282 (Insert Fig. 4a, 4b)

283

284 *3.3. Storage of soil nitrogen*

285 Total N was found in 48 studies with a total sample size of 167 RR values. The estimated RR
286 values from each study and the overall means are shown in Figures 5a. The overall mean effect size (RR
287 = 1.13; CL: 1.08-1.19) was significantly greater than 1 indicating that soil N stocks under agroforestry
288 were higher than in crop monocultures. The effect sizes in simultaneous systems did not significantly
289 differ from the sequential systems (Figure 5b). The difference between N-fixing and non N-fixing species
290 was also not statistically significant. Hence, our hypothesis that non N-fixing agroforestry trees
291 contribute to greater soil N stock build up was not supported. The effect of agroforestry on soil total N
292 levels was significantly influenced by soil texture (Figure 5b). Total N was significantly higher in sandy
293 soils than loamy soils. Aggregate-associated N was significantly higher under agroforestry than in the
294 crop monocultures (Figure 5b). Closer examination using soil physical fractionation techniques shows
295 that 22-43% greater soil N is stored in macroaggregates under agroforestry practices.

296

297 (Insert Fig. 5a, 5b)

298

299 *3.4. Availability of soil nitrogen*

300 Data on available N were found in 34 studies with a total of 117 RR values. Figure 6a gives the
301 estimated values of effect sizes for each study and the overall mean. The overall mean RR (1.46; CL:
302 1.32-1.59) was significantly greater than 1 indicating that soil available N under agroforestry was 46%
303 higher than in crop monocultures (Figure 6a). The increase in soil N availability was most readily
304 detected as nitrate-N rather than as ammonium-N. Soil N availability did not significantly vary with
305 agroforestry management, ability to fix N or soil texture (Figure 6b). However, agroforestry increased
306 available soil N by up to 52% on clay soils as compared to the 25% increase on loamy soils (Figure 6b).

307 (Insert Fig. 6a, 6b)

308

309 *3.5. Availability of soil phosphorus*

310 Soil available P was found in 49 studies with a total sample size of 165 RR values. Variations in
311 RR with study and the overall (all studies) effect size are presented in Figure 7a. The overall mean RR
312 was 1.11 (CL: 1.05-1.68) was significantly greater than 1. However, the increase due to agroforestry
313 practices was marginal in most studies (Figure 7a). No significant differences were found between
314 sequential and simultaneous systems or N fixing and non N-fixing tree species (Figure 7b). P availability
315 was significantly higher on loamy soils than sandy soils (Figure 7b).

316

317 (Insert Fig. 7a, 7b)

318

319 *3.5. Alleviation of soil acidity*

320 Soil pH was found in 46 studies with a total sample size of 138 RR values. Figure 8a shows the
321 variations in RR with study and across all studies. Overall, agroforestry practices significantly increased

322 soil pH (RR = 1.02; CL: 1.01-1.03) over the crop monoculture. However, the effect sizes did not
323 significantly differ with agroforestry practice, the ability of trees to fix N or soil texture (Figure 8b). RR
324 values greater than 1 were found in pH below 6, while above pH 7 the RR values remained close to 1
325 (Figure 9a). The effect of agroforestry on soil pH also marginally differed with soil type; the most
326 significant increase in pH being on Nitisols, Ferralsols and Acrisols (Figure 9b), which are naturally
327 prone to acidification.

328

329 (Insert Fig. 8a, 8b, 9a, 9b)

330

331 **4. Discussion**

332 *4.1. Agroforestry reduces erosion rates*

333 Soil erosion is one of the most pervasive features of land degradation globally (FAO and ITPS,
334 2015), and erosion by water is particularly widespread in mountainous agricultural landscapes in humid
335 tropical and sub-tropical regions (Labrière et al., 2015). Soil erosion has numerous on-site and off-site
336 impacts. On-site impacts result in decline in soil quality because of the loss of key soil constituents (e.g.,
337 SOC, clay, and silt), reduction in water holding capacity and nutrient reserves, loss of topsoil where most
338 soil organic matter and soil organisms are found, and decline in the efficient use of inherent and applied
339 nutrients.

340 Our analysis provides evidence supporting the hypothesis that agroforestry practices significantly
341 reduce soil erosion rates compared to crop monocultures in humid and sub-humid tropics. This
342 conclusion is supported by the reduction in erosion rates, higher infiltration rates and macroaggregation,
343 and lower runoff recorded under agroforestry. Following the conversion of natural vegetation to
344 agricultural land, soil erosion is often increased due to removal of the litter layer protecting the soil as
345 well as tillage practices (Montgomery 2007; Labrière et al., 2015). The provision of organic inputs by

346 agroforestry trees through litterfall and prunings contributes to soil cover. Trees can also provide physical
347 barriers to soil erosion (Angima et al., 2002). This combined with the predominance of reduced/no-tillage
348 practices in agroforestry (Barrios et al., 2012) is likely an important reason for the lower soil erosion
349 rates. Furthermore, the belowground organic inputs through root turnover and the increased biological
350 activity of soil ecosystem engineers (Pauli et al., 2010; Kamau et al 2017) that promote soil structural
351 stability are important contributors to the reduction in soil erosion rates under agroforestry (Six et al.,
352 2002; Fonte et al., 2010). The abundance of large macroaggregates and MWD under agroforestry could
353 also partly explain the reduction in erosion rates. Large relative values of macroaggregates and MWD
354 indicate that aggregate forming processes predominate over aggregate destroying factors and thus soil
355 structure is being consolidated and net soil erosion reduced (Six et al., 1998). Stable aggregates are built
356 by biological activity, and largely bound together by fungal hyphae, and plant and microbial exudates
357 that bind soil particles together. Aggregate stability therefore is an important indicator of the structural
358 stability of soil and its resistance to erosion.

359 The data did not support our hypothesis that agroforestry contributes to lower erosion rates on fine-
360 textured soils than coarser textured soils. Medium-textured soils having a high silt content are often said
361 to be the most erodible of all soils (FAO and ITPS, 2015). However, this analysis did not reveal
362 significant differences among soil texture classes. These results together provide evidence that
363 agroforestry can play a vital role in erosion control, which is one of the key regulation services in
364 agroecosystems. As such it can reduce on-site and off-site impacts of conventional agricultural practices
365 and inputs (e.g. fertilizers, biocides and other toxic chemicals).

366

367 *4.2. Agroforestry increases storage of SOC*

368 Our results provide evidence supporting the hypothesis that agroforestry contributes to greater SOC
369 build-up than crop monocultures. All agroforestry systems studied had a similar contribution to increased

370 SOC levels and this effect is consistent with increased soil aggregation levels as part of soil structure
371 improvement. This is because soil C is protected inside soil aggregates (Six et al., 2002; Fonte et al.,
372 2010) leading to as much as 30% greater soil C stored in soil macroaggregates under agroforestry
373 practices which is consistent with other studies reported in the literature (Guo and Gifford, 2002). The
374 increase in SOC storage (and hence SOM) has significant implications for provisioning (e.g., increased
375 crop productivity) as well as regulating (e.g., carbon sequestration, soil erosion control) ecosystem
376 services (Barrios, 2007; Palmer et al., 2017). At the farm scale, not only does retaining high SOM affect
377 nutrient availability and growth of crop plants, but also soil biodiversity and bottom-up effects on crop
378 pests and their natural enemies (Scheu, 2001; Veen et al., 2019). For example, high SOM content in soil
379 can support a greater diversity of soil organisms, which provide alternative food sources for natural
380 enemies that help to suppress crop pests (Scheu, 2001). The SOM is also a source of food for termites,
381 which become a problem in cropping systems with low SOM (Sileshi et al., 2005). SOC also affects
382 multiple soil physical properties including aggregate stability, bulk density and water infiltration rates.
383 Interestingly, a recent meta-analysis by Minasny and McBratney (2018) highlights that contribution to
384 the overall increase in available water capacity seems to be lower than commonly thought as 1% mass
385 increase in SOC on average increased available water capacity by about 1.2%. In contrast, even small
386 changes in SOC stock can have considerable impacts on the atmospheric CO₂ concentrations and the
387 global climate (Paustian et al., 2016).

388

389 *4.3 Agroforestry increases storage and availability of soil N*

390 Our results support the hypothesis that agroforestry significantly contributes to greater soil total N
391 levels than crop monocultures. Since most soil N is found in organic form as part of SOM, N follows a
392 similar distribution and dynamics to that of SOC (Barrios et al, 2012; Weil and Brady, 2017). Hence,
393 SOM protection inside aggregates is an important mechanism for N storage in soil.

394 Our hypothesis that agroforestry contributes to greater soil available N than agriculture without
395 trees was supported by the data. N availability largely controls the net primary production and
396 functioning of both managed and natural ecosystems (Chapin et al. 2011). Although significantly smaller
397 in size, the soil available N pool (which is largely constituted of ammonium-N and nitrate-N) is more
398 readily impacted by land use and management than soil total N (Barrios et al., 1996). Nevertheless,
399 agroforestry trees with higher organic tissue quality (i.e. lower C/N, L/N and L+PP/N ratios) and faster
400 decomposition rates have been shown to make greater short-term contributions to soil available N than
401 trees with lower tissue quality (Barrios et al., 1997; Cobo et al., 2002; Vanlauwe et al., 2005a).

402 *4.4 Agroforestry increases soil available P*

403 Phosphorus availability is a widespread nutrient constraint to net primary productivity and crop
404 production in tropical and subtropical soils (Vitousek et al., 2010). Furthermore, Soil P stocks are also
405 declining in large regions of the world due to greater export of P through removal of harvested products
406 and erosion than input of P to soils (Sanchez et al. 2019).

407 Our results support the hypothesis that agroforestry significantly increases soil available P
408 compared with crop monocultures. The possible mechanisms for improved P availability in agroforestry
409 include (a) the mineralization of organically bound P in the organic inputs; (b) the transformation of less
410 available pools of inorganic P into more readily available organic P that is mineralized, when plants
411 convert inorganic P in their tissues, and those are cycled back to the soil; and (c) organic C radicals
412 blocking P-sorption sites (Sanchez et al., 1997). In addition, many tree species used in agroforestry
413 systems are highly depended on arbuscular mycorrhizal fungi (AMF) and cluster roots to adapt to P-
414 deficient soils (Lambers et al., 2008; Bainard et al., 2011). AMF play a critical role in the uptake of
415 relatively immobile forms of P through their effects on increased mobilization of P in the rhizosphere
416 (Radersma and Grierson, 2004; Carvalho et al. 2010). For example, *T. diversifolia* is highly mycorrhizal
417 (Sharrock et al., 2004) and has been shown to accumulate P-rich biomass (Jama et al., 2000; Barrios and

418 Cobo, 2004). Its application to P-fixing soils in the Colombian Andes resulted in increased labile soil
419 organic P and soil available P (Phiri et al., 2001). Similarly, *Sesbania sesban* fallows increased labile soil
420 organic P for three consecutive post-fallow seasons in Western Kenya (Maroko et al., 1999).

421 The hypothesis that sequential agroforestry practices contribute to greater soil P availability than
422 simultaneous practices was not supported in this analysis. Similarly, N-fixers and non-N fixers
423 contributed to soil available P equally. Agroforestry also significantly increased soil available P over the
424 control in sandy soils but not in clay and loam soils. Overall, the results provide evidence that
425 agroforestry can lead to increases in P availability although the increases are marginal. While
426 agroforestry trees can enhance P cycling in particular contexts by mobilizing less-available forms of soil
427 P into more readily available organic P pools in the soil, the strategic inputs of P fertilizer are still
428 necessary to increased and sustained agricultural production in low-P soils (Rao et al. 2004).

429 *4.5. Agroforestry alleviates soil acidity*

430 Inherent soil acidity due to parent material and soil acidification are recognized as important
431 limitations to agricultural intensification (Guo et al., 2010, FAO and ITPS, 2015). In tropical and sub-
432 tropical regions, the impact of high rainfall on leaching of base cations, the predominant application of
433 ammonium-based fertilizers and the removal of base cations by plants and crop offtake have been
434 identified as major contributors to increased soil acidity (FAO and ITPS, 2015). Soil acidity leads to
435 nutrient deficiencies and toxicities besides negatively affecting activities of beneficial microorganisms,
436 decomposition of organic matter, nutrient mineralization and crop uptake (FAO and ITPS, 2015). Our
437 results support the hypothesis that agroforestry can contribute to alleviating soil acidity compared to
438 crop monocultures. However, our results do not support the notion that N fixing trees increase soil acidity
439 (McLay et al., 1997). This is consistent with a recent review showing that N-fixing trees can contribute
440 to reduce soil acidification (Sileshi et al., 2014). Tropical legumes typically take up less cations and have

441 lower acidifying effect on the rhizosphere because the amino acids produced by N-fixation have lower
442 propensity to release protons (Bohan et al. 1991).

443 Trees could minimize soil acidification both by decreasing drainage and through deep-capture and
444 recycling leached nutrients. However, the soil acidity alleviating effect of plant materials depends on
445 their chemical composition, especially their ash alkalinity (Wong et al. 2002). For example, *Senna*
446 *siamea* has been shown to recycle Ca from subsoils and significantly increase pH in the top soil
447 (Vanlauwe et al., 2005b). Trees producing litter rich in Ca are often associated with soils with higher
448 exchangeable Ca, per cent base saturation and pH (Dijkstra, 2003; Reich et al., 2005). The concentration
449 of Ca in soil influences soil pH because it is a base cation that competes with cations promoting acidity
450 for exchange sites on soil particle surfaces and organic matter (Weil and Brady, 2017). Increases in soil
451 pH have often been associated with greater abundance and activity of soil organisms that can influence
452 C and nutrient cycling (Reich et al., 2005).

453

454 **5. Conclusions**

455 This analysis has demonstrated that agroforestry can significantly reduce soil erosion rates,
456 increase SOC and N storage, increase the availability of N and marginally increase available P and pH
457 in the soil compared to crop monocultures. As such, agroforestry can be an option for increasing soil
458 nutrient availability to crops when access or use of mineral fertilizers is limited. Furthermore, by
459 facilitating the combined application of organic and mineral nutrient inputs to soil, agroforestry can
460 significantly improve nutrient use efficiency through greater synchronization of nutrient release to soil
461 and crop demand and use.

462 We conclude that agroforestry can significantly contribute to ecological intensification trajectories
463 that support agroecological transitions towards sustainable food and agricultural systems in the humid
464 and sub-humid tropics. It can also provide significant climate change adaptation and mitigation benefits.

465 Therefore, we recommend that agroforestry be considered in the nationally determined contributions of
466 parties to the UNFCCC in the coming years.

467

468 **Author contributions**

469 All authors jointly designed the research; MNM and EB compiled data. GWS analyzed data with inputs
470 from EB, MNM, SG, MJ and LP. GWS and EB wrote the draft manuscript with inputs from MNM,
471 SG, MJ and LP and SC.

472

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477

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