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 macroggregates, 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

1. Introduction

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

Two major types of agroforestry practices can be distinguished: i) simultaneous agroforestry where trees and crops occur on the same piece of land during the same cropping season (e.g. shaded coffee and cocoa systems, alley cropping, intercropping), and ii) sequential agroforestry where trees and crops occur 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 heath 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

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

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

2.2. Research questions and hypotheses

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

(insert Table 1)

2.3. Literature search

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

2.4. Criteria for inclusion in meta-analysis

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

1. The study originated in the humid or sub-humid tropics (annual rainfall >600 mm, within 30° 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

9

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 overal 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

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

(Insert Fig. 4a, 4b)

3.3. Storage of soil nitrogen

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

(Insert Fig. 5a, 5b)

3.4. Availability of soil nitrogen

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

(Insert Fig. 6a, 6b)

3.5. Availability of soil phosphorus

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

(Insert Fig. 7a, 7b)

3.5. Alleviation of soil acidity

Soil pH was found in 46 studies with a total sample size of 138 RR values. Figure 8a shows the 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

16

SOC levels and this effect is consistent with increased soil aggregation levels as part of soil structure improvement. This is because soil C is protected inside soil aggregates (Six et al., 2002; Fonte et al., 2010) leading to as much as 30% greater soil C stored in soil macroaggregates under agroforestry practices which is consistent with other studies reported in the literature (Guo and Gifford, 2002). The increase in SOC storage (and hence SOM) has significant implications for provisioning (e.g., increased crop productivity) as well as regulating (e.g., carbon sequestration, soil erosion control) ecosystem services (Barrios, 2007; Palmer et al., 2017). At the farm scale, not only does retaining high SOM affect nutrient availability and growth of crop plants, but also soil biodiversity and bottom-up effects on crop pests and their natural enemies (Scheu, 2001; Veen et al., 2019). For example, high SOM content in soil can support a greater diversity of soil organisms, which provide alternative food sources for natural enemies that help to suppress crop pests (Scheu, 2001). The SOM is also a source of food for termites, which become a problem in cropping systems with low SOM (Sileshi et al., 2005). SOC also affects multiple soil physical properties including aggregate stability, bulk density and water infiltration rates. Interestingly, a recent meta-analysis by Minasny and McBratney (2018) highlights that contribution to the overall increase in available water capacity seems to be lower than commonly thought as 1% mass increase in SOC on average increased available water capacity by about 1.2%. In contrast, even small changes in SOC stock can have considerable impacts on the atmospheric CO_2 concentrations and the global climate (Paustian et al., 2016).

4.3 Agroforestry increases storage and availability of soil N

Our results support the hypothesis that agroforestry significantly contributes to greater soil total N levels than crop monocultures. Since most soil N is found in organic form as part of SOM, N follows a similar distribution and dynamics to that of SOC (Barrios et al, 2012; Weil and Brady, 2017). Hence, 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 rations) 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 contributes 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.

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