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ABSTRACT

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

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

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

(Insert Fig. 1)

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

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

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

Reviews and meta-analyses published in the last three decades have increased our understanding of the impact of agroforestry on some of the provisioning services such as crop yields (e.g. Bayala et al., 2012; Kuyah et al., 2016; Sileshi et al., 2008), and regulating services such as control of pests, diseases and weeds (e.g. Pumariño et al., 2015) and carbon sequestration (e.g. Chatterjee et al., 2018, Cerda et al. 2019). However, similar reviews and meta-analysis do not exist on the mechanisms by which agroforestry practices impact on soil health and soil-mediated ecosystem services. This synthesis was designed to address research questions and hypotheses (Table 1) that have remained outstanding and were not addressed by earlier syntheses and meta-analyses. Although several studies assessing the effects of agroforestry on various soil properties have been published in the last three decades, a quantitative synthesis of the results from those studies is still lacking. Therefore, the objective of this meta-analysis is to quantify the contribution of agroforestry practices to soil-mediated ecosystem services, specifically, regulation of soil erosion, storage of soil carbon (C), storage of soil nitrogen (N), availability of soil N, availability of soil phosphorus (P), and alleviation of soil acidity across the humid and sub-humid tropics. The overall aim of this synthesis is to create awareness among researchers, development practitioners and policy-makers on the roles that agroforestry can play in climate change adaptation and mitigation as well as management of land degradation. This kind of information is hoped to be useful as countries engage in the Koronivia Joint Work on Agriculture and the preparation of their next nationally determined contributions (NDCs) to the UNFCCC.

2. Methods

2.1. Selection of indicators

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

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

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

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

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

2.5. Data extraction

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

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

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

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

2.6. Effect size

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

2.7. Data analysis

We applied a linear mixed modelling procedure for all analyses. We preferred the mixed modelling approach because many of the studies did not report either the SD or SE. The mixed modelling procedure was also more appropriate as the data gathered across studies were unbalanced with respect to predictor variables and sample sizes. In the mixed model we entered the categorical variables (e.g. agroforestry type, ability to fix N and soil texture) as fixed effects and the source of data (i.e. study) as the random effect. Then we estimated model parameters and their 95% confidence intervals (95% CI) using restricted maximum likelihood estimation (REML). Where moderator variables were not applicable, for example the overal effect of agroforestry on a given variable, the 95% CIs were estimated by bootstrapping (resampling with replacement) with 9999 random replicates.

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

3. Results

3.1. Regulation of soil erosion

Erosion rate were reported in a total of 17 studies and a sample size of 69 was available for analysis. The estimated effect sizes in each study and the overall (all studies combined) are presented in Figures 2a. In all studies, RR was less than 1 indicating that soil erosion rates were significantly lower under agroforestry compared to the corresponding crop monocultures. Overall, agroforestry trees reduced soil erosion by 50% (Figure 2a). All the studies on soil erosion were conducted in simultaneous agroforestry, and only one study was found on sequential systems. Although the differences were not statistically significant (Figure 2b), tree species that do not fix N generally contributed to lower erosion (RR = 0.29) than N-fixing trees (RR = 0.41). The effect sizes for erosion rates also did not significantly differ among soil texture classes, but loamy soils had generally lower erosion rates than sandy soils (Figure 2b).

(Insert Fig. 2a, 2b)

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

(Insert Fig. 3a, 3b)

3.2. Storage of soil carbon

SOC was reported in 71 studies and a total of 225 pairs of observations were available for analysis. The estimated effect sizes for each study and the overall mean RR are presented in Figure 4a. With overall effect size of 1.21 (CL: 1.15-1.27), agroforestry significantly increased SOC storage compared to crop monocultures although effects varied with study (Figure 4a). However, the effect size did not 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 soil pH (RR = 1.02; CL: 1.01-1.03) over the crop monoculture. However, the effect sizes did not significantly differ with agroforestry practice, the ability of trees to fix N or soil texture (Figure 8b). RR values greater than 1 were found in pH below 6, while above pH 7 the RR values remained close to 1 (Figure 9a). The effect of agroforestry on soil pH also marginally differed with soil type; the most significant increase in pH being on Nitisols, Ferralsols and Acrisols (Figure 9b), which are naturally prone to acidification.

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

4. Discussion

4.1. Agroforestry reduces erosion rates

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

Our analysis provides evidence supporting the hypothesis that agroforestry practices significantly reduce soil erosion rates compared to crop monocultures in humid and sub-humid tropics. This conclusion is supported by the reduction in erosion rates, higher infiltration rates and macroaggregation, and lower runoff recorded under agroforestry. Following the conversion of natural vegetation to agricultural land, soil erosion is often increased due to removal of the litter layer protecting the soil as well as tillage practices (Montgomery 2007; Labrière et al., 2015). The provision of organic inputs by agroforestry trees through litterfall and prunings contributes to soil cover. Trees can also provide physical barriers to soil erosion (Angima et al., 2002). This combined with the predominance of reduced/no-tillage practices in agroforestry (Barrios et al., 2012) is likely an important reason for the lower soil erosion rates. Furthermore, the belowground organic inputs through root turnover and the increased biological activity of soil ecosystem engineers (Pauli et al., 2010; Kamau et al 2017) that promote soil structural stability are important contributors to the reduction in soil erosion rates under agroforestry (Six et al., 2002; Fonte et al., 2010). The abundance of large macroaggregates and MWD under agroforestry could also partly explain the reduction in erosion rates. Large relative values of macroaggregates and MWD indicate that aggregate forming processes predominate over aggregate destroying factors and thus soil structure is being consolidated and net soil erosion reduced (Six et al., 1998). Stable aggregates are built by biological activity, and largely bound together by fungal hyphae, and plant and microbial exudates that bind soil particles together. Aggregate stability therefore is an important indicator of the structural stability of soil and its resistance to erosion.

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

4.2. Agroforestry increases storage of SOC

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

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.

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

4.4 Agroforestry increases soil available P

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

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

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

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

4.5. Agroforestry alleviates soil acidity

Inherent soil acidity due to parent material and soil acidification are recognized as important limitations to agricultural intensification (Guo et al., 2010, FAO and ITPS, 2015). In tropical and sub-tropical regions, the impact of high rainfall on leaching of base cations, the predominant application of ammonium-based fertilizers and the removal of base cations by plants and crop offtake have been identified as major contributors to increased soil acidity (FAO and ITPS, 2015). Soil acidity leads to nutrient deficiencies and toxicities besides negatively affecting activities of beneficial microorganisms, decomposition of organic matter, nutrient mineralization and crop uptake (FAO and ITPS, 2015). Our results support the hypothesis that agroforestry can contributes to alleviating soil acidity compared to crop monocultures. However, our results do not support the notion that N fixing trees increase soil acidity (McLay et al., 1997). This is consistent with a recent review showing that N-fixing trees can contribute to reduce soil acidification (Sileshi et al., 2014). Tropical legumes typically take up less cations and have lower acidifying effect on the rhizosphere because the amino acids produced by N-fixation have lower propensity to release protons (Bohan et al. 1991).

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

5. Conclusions

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

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

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

Author contributions

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

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References

Akinnifesi, F.K., Ajayi, O.C., Sileshi, G., Chirwa, P.W., Chianu, J., 2010. Fertiliser trees for sustainable food security in the maize-based production systems of East and Southern Africa. A review. Agron. Sust. Dev. 30, 615–629.

Angima, S.D., Stott D.E., O'Neill M.K., Ong C.K., Weesies G.A., 2002. Use of calliandra–Napier grass contour hedges to control erosion in central Kenya. Agric. Ecosyst. Environ. 91, 15–23.

Bainard L.D., Klironomos J.N., Gordon A.M., 2011. Arbuscular mycorrhizal fungi in tree-based intercropping systems: A review of their abundance and diversity. Pedobiologia 54, 57–61.

Barrios, E., 2007. Soil biota, ecosystem services and land productivity. Ecol. Econ. 64, 269–285.

Barrios, E., Buresh, R.J., Sprent, J.I., 1996a. Nitrogen mineralization in density fractions of soil organic matter from maize and legume cropping systems. Soil Biol. Biochem. 28, 1459–1465.

Barrios, E., Kwesiga, F., Buresh, R.J., Sprent, J.I., 1997. Light fraction soil organic matter and available nitrogen following trees and maize. Soil Sci. Soci. Amer. J. 61, 826–831.

Barrios, E., Kwesiga, F., Buresh, R.J., Sprent, J.I., Coe, R., 1998. Relating preseason soil nitrogen to maize yield in tree legume-maize rotations. Soil Sci. Soci. Amer. J. 62, 1604–1609.

Barrios, E., Cobo, J.G., 2004. Plant growth, biomass production and nutrient accumulation by slash/mulch agroforestry systems in tropical hillsides of Colombia. Agrofor. Syst. 60, 255–265.

Barrios, E., Cobo, J.G., Rao, I.M., Thomas, R.J., Amezquita, E., Jimenez, J.J., Rondon, M.A., 2005. Fallow management for soil fertility recovery in tropical Andean agroecosystems in Colombia. Agric. Ecosyst. Environ. 110, 29–42.

Barrios, E., Sileshi G.W., Shepherd K., Sinclair F., 2012. Agroforestry and soil health: linking trees, soil biota and ecosystem services. In: Wall, D.H. et al. (Eds.). Soil Ecology and Ecosystem Services. First Edition. Oxford University Press, Oxford, UK. Pp. 315–330.

Barrios, E., Shepherd, K.; Sinclair, F. 2015. Soil health and agricultural sustainability: the role of soil biota. In FAO. Agroecology for Food Security and Nutrition: Proceedings of the FAO International Symposium, pp. 104-122. Rome.

Barrios, E., Valencia, V., Jonsson, M., Brauman, A., Hairiah, K., Mortimer, P., Okubo, S., 2018. Contribution of trees to the conservation of biodiversity and ecosystem services in agricultural landscapes. Int. J. Biodivers. Sci. Ecosyst. Serv. Manage. 14, 1–16.

Bohan, N.S., Herdley, M.J., White, R.E., 1991. Processes of soil acidification during nitrogen cycling with emphasis on legume based pastures. Plant Soil 134, 53-63.

Buresh, R.J., Tian, G., 1998. Soil improvement by trees in sub-Saharan Africa. Agroforestry Systems 38, 51-76.

Carvalho, A.M.X., Tavares, R.C., Cardoso, I.M., Kuyper, T.W., 2010. Mycorrhizal associations in agroforestry systems. In P. Dion (Ed.). Soil Biology and Agriculture in the Tropics, Springer-Verlag, Heidelberg, Germany. Pp. 185-208.

Cerda, R., Orozco-Aguilar, L., Sepulveda, N., Ordonez, J., Carreno-Rocabado G., 2019. Tropical agroforestry and ecosystems services: trade-off analysis for better design strategies. In Mosquera-Losada, M.R., Prabhu, R. Eds.) Agroforestry for sustainable agriculture. Burleigh Dodds Science Publishing, Cambridge, UK.

Chapin, F.S.; Matson, P.A.; Vitousek, P., 2011. Principles of terrestrial ecosystem ecology - 2nd Edition. Springer, New York. 528 pp.

Chatterjee, N., Naira, P.K.R., Chakraborty S., Nair, V.D., 2018. Changes in soil carbon stocks across the forest-agroforest-agriculture/pasture continuum in various agroecological regions: A meta-analysis. Agric. Ecosyst. Environ. 266, 55–67.

Cobo, J.G., Barrios, E., Kass, D., Thomas, R.J., 2002. Decomposition and nutrient release by green manures in a tropical hillside agroecosystem. Plant Soil 240, 331–342.

Dijkstra, F.A., 2003. Calcium mineralization in the forest floor and surface soil beneath different tree species in the northeastern US. Forest Ecol. Manage. 175, 185–194.

Doran, J.W., 2001. Soil health and global sustainability: Translating science in to practice. Agr. Ecosyst. Environ. 1826, 1–9.

FAO, 2017. The future of food and agriculture – Trends and challenges. Rome, Italy.

FAO and ITPS, 2015. Status of the world's soil resources (SWSR) – Main Report. Food and Agriculture Organization of the United Nations and Intergovernmental Technical Panel on Soils, Rome, Italy.

FAO, 2018. The Koronivia joint work on agriculture and the convention bodies: an overview. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Fonte, S.J., Barrios, E., Six, J., 2010. Earthworms, soil fertility and aggregate-associated soil organic matter dynamics in the Quesungual agroforestry system. Geoderma 155, 320–328.

Garratt, M.P.D., Bommarco, R., Kleijn, D., Martin, E., Mortimer, S.R., Redlich, S., Senapathi, D., Steffan-Dewenter, I. Świtek, S., Takács, V., van Gils, S., van der Putten, W.H., Potts, S.G., 2018. Enhancing soil organic matter as a route to the ecological intensification of European arable systems. Ecosystems 21, 1404–1415.

Gliessman, SR., 2007. Agroecology: The ecology of sustainable food systems. Boca Raton, FL: CRC Press. 384 pp.

Guo L.B., Gifford R.M., 2002. Soil carbon stocks and land use change: a meta-analysis. Global Change Biology 8, 345-360.

Guo, J.H., Liu X.J., Zhang Y., Shen J.L., Han W.X., Zhang W.F., Christie P., Goulding K.W.T., Vitousek P.M., Zhang F.S., 2010. Significant acidification in major Chinese croplands. Science 327, 1008–1010.

Hedges, L.V., Gurevitch, J., Curtis, P.S., 1999. The meta-analysis of response ratios in experimental ecology. Ecology 80, 1150–1156.

Heink, U., Kowarik, I., 2010. What are indicators? On the definition of indicators in ecology and environmental planning. Ecol. Indic. 10, 584–593.

Heinimann, A., Mertz, O., Frolking, S., Christensen, A.E., Hurni, K., Sedano, F., et al., 2017. A global view of shifting cultivation: Recent, current, and future extent. PLoS ONE 12, e0184479.

Jama, B., Palm, C.A., Buresh, R.J., Niang, A., Gachengo, C., Nziguheba, G., Amadalo, B., 2000. *Tithonia diversifolia* as a green manure for soil fertility improvement in western Kenya: a review. Agrofor. Syst. 49, 201–221.

Johnson, D.W.; Curtis, P.S., 2001. Effects of forest management on soil C and N storage: meta-analysis. Forest Ecol. Mgmt.140, 227-238.

Kamau, S., Barrios, E., Karanja, N., Ayuke, F., Lehmann, J., 2017. Soil macrofauna under dominant tree species increases along a soil degradation gradient. Soil Biol. Biochem. 112, 35–46.

Kibblewhite, M.G.; Ritz, K.; Swift, M.J., 2008. Soil health in agricultural systems. Phil. Trans. Royal. Soc. B 363, 685–701.

Kuyah, S., Öborn, I., Jonsson, M., Dahlin, A.S., Barrios, E., Muthuri, C., Malmer, A., Nyaga, J., Magaju, C., Namirembe, S., Nyberg, Y., Sinclair, F.L., 2016. Trees in agricultural landscapes enhance provision of ecosystem services in Sub-Saharan Africa. Int. J. Biodivers. Sci. Ecosyst. Servic. Manage. 12, 255–273.

Labrière, N., Locatelli, B., Laumonier, Y., Freycon, V., Bernoux, M., 2015. Soil erosion in the humid tropics: A systematic quantitative review. Agric. Ecosyst. Environ. 203, 127–139.

Lal, R., 2015. Restoring soil quality to mitigate soil degradation. Sustainability 7, 5875–5895.

Lal, R., 2016. Feeding 11 billion on 0.5 billion hectare of area under cereal crops. Food Energy Secur. 5, 239–251.

Lambers, H., Raven, J.A., Shaver, G.R., Smith, S.E., 2008. Plant nutrient-acquisition strategies change with soil age. Trends Ecol. Evol. 23, 95–103.

Maroko, J.B., Buresh R.J., Smithson P.C. 1999. Soil phosphorus fractions in unfertilized fallow-maize systems on two tropical soils. Soil Sci. Soc. Amer. J. 63, 320–326.

McLay, C.D.A., Barton, L., Tang, C., 1997. Acidification potential of ten grain legume species grown in nutrient solution. Aus. J. Agric. Res. 48, 1025–1032.

Minasny, B. and McBratney, A.B., 2018. Limited effect of organic matter on soil available water capacity. Eur. J. Soil Sci. 69, 39–47Montgomery, D.R., 2007. Soil erosion and agricultural sustainability. Proc. Nat. Acad. Sci. USA 104, 13268–13272.

Mutuo, P.K., Cadisch, G., Albrecht, A., Palm, C.A., Verchot, L., 2005. Potential of agroforestry for carbon sequestration and mitigation of greenhouse gas emissions from soils in the tropics. Nutr. Cycl. Agroecosyst. 71, 43–54.

Myers, N., Mittermeler, R.A., Mittermeler, C.G., Da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858.

Nair, P.K., Garrity, D., 2012. Agroforestry – The future of global land use. Advances in Agroforestry Book Series. Springer, Germany.

O'Neill, D.W., Fanning, A.L., Lamb, W.F., Steinberger, J.K., 2018. A good life for all within planetary boundaries. Nature Sust. 1, 88–95.

Palmer, J., Thorburn, P.J., Biggs, J.S., Dominati, E.J., Probert, M.E., Meier, E.A., Huth, N.I., Dodd, M., Snow, V., Larsen, J.R., Parton, W.J., 2017. Nitrogen cycling from increased soil organic carbon contributes both positively and negatively to ecosystem services in wheat agro-ecosystems. Front. Plant Sci. 8, 731. doi: 10.3389/fpls.2017.00731

Pauli, N., Oberthur, T., Barrios, E., Conacher, A., 2010. Fine-scale spatial and temporal variation in earthworm surface casting activity in agroforestry fields, western Honduras. Pedobiologia 53, 127–139.

Paustian, K., Lehmann, J., Ogle, S., Reay, D., Robertson, G.P., Smith, P., 2016. Climate-smart soils. Nature 532, 49–57.

Phiri, S., Barrios E., Rao, I.M., Singh, B.R., 2001. Changes in soil organic matter and phosphorus fractions under planted fallows and a crop rotation system on a Colombian volcanic-ash soil. Plant Soil 231, 211–223.

Prabhu, R.; Barrios, E.; Bayala, J.; Diby, L.; Donovan, J.; Gyau, A.; Graudal, L.; Jamnadass, R.; Kahia, J.; Kehlenbeck, K.; Kindt, R.; Kouame, C.; McMullin, S.; van Noordwijk, M.; Shepherd, K.; Sinclair, F.; Vaast, P.; Vågen, T.-G.; Xu, J., 2015. Agroforestry: realizing the promise of an agroecological approach. In FAO. Agroecology for Food Security and Nutrition: Proceedings of the FAO International Symposium, pp. 201-224. Rome.

Pretty J.N., 2018. Intensification for redesigned and sustainable agricultural systems. Science 362 (3417), 1-7.

Pumariño, L., Sileshi, G.W., Gripenberg, S., Kaartinen, R., Barrios, E., Muchane, M.N., Midega, C., Jonsson, M., 2015. Effects of agroforestry on pest, disease and weed control: a meta-analysis. Basic Appl. Ecol. 16, 573–582.

Radersma, S., Grierson P.F., 2004. Phosphorus mobilization in agroforestry: Organic anions, phosphatase activity and phosphorus fractions in the rhizosphere. Plant Soil 259, 209–219.

Reich, P.B., Oleksyn, J., Modrzynski, J., Mrozinski, P., Hobbie, S.E., Eissenstat, D.M., Chorover, J., Chadwick, O.A., Hale, C.M., Tjoelker M.G., 2005. Linking litter calcium, earthworms and soil properties: a common garden test with 14 tree species. Ecol. Lett. 8, 811–818.

Robinson, D. A., Fraser, I., Dominati, E. J., Davíðsdóttir, B., Jónsson, J. O. G., Jones, L., et al., 2014. On the value of soil resources in the context of natural capital and ecosystem service delivery. Soil Sci. Soc. Am. J. 78, 685–700.

Rutgers, M., van Wijnen, H.J., Schouten, A.J., Mulder, C., Kuiten, A.M.P., Brussaard, L., Breure A.M., 2012. A method to assess ecosystem services developed from soil attributes with stakeholders and data of four arable farms. Sci. Total Environ. 415, 39–48.

Sanchez, P.A., Buresh, R.J., Leakey, R.R.B. 1997. Trees, soils, and food security. Phil. Trans. R. Soc. Lond. B 352, 949-961.

Sanchez, P.A., 2019. Properties and management of soils in the tropics – 2nd Edition. Cambridge University Press, Cambridge, U.K.

Scheu, S., 2001. Plants and generalist predators as links between the below-ground and above-ground system. Basic Appl. Ecol. 2, 3–13.

Schoeneberger, M., Bentrup, G., de Gooijer, H., Soolanayakanahally, R., Sauer, T., Brandle, J., Zhou, X., Current, D., 2012. Branching out: Agroforestry as a climate change mitigation and adaptation tool for agriculture. J. Soil Water Conserv. 67, 128A–136A.

Sharrock, R.A., Sinclair, F.L., Gliddon, C., Rao, I.M., Barrios, E., Smithson, P., Jones, D.L., Godbold, D.L., 2004. A global assessment using PCR techniques of mycorrhizal fungal populations colonising *Tithonia diversifolia*. Mycorrhiza 14, 103–109.

Shirazi, M.A., Boersma, L., 1984. A unifying quantitative analysis of soil texture. Soil. Sci. Soc. Am. J. 48, 142–147.

Sileshi, G., Mafongoya, P.L., Kwesiga, F., Nkunika, P., 2005. Termite damage to maize grown in agroforestry systems, traditional fallows and monoculture on Nitrogen-limited soils in eastern Zambia. Agric. For. Entomol. 7, 61–69.

Sileshi, G.W., Mafongoya, P.L., Akinnifesi, F.K., Phiri, E., Chirwa, P., Beedy, T., Makumba, W., Nyamadzawo, G., Njoloma, J., Wuta, M., Nyamugafata, P., Jiri, O., 2014. Fertilizer Trees. *Encyclopedia of Agriculture and Food Systems,* Vol. 1, San Diego: Elsevier; pp. 222–234.

Six, J., Elliot, E.T., Paustian, K., Doran, J.W., 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. Soil Sci. Soci. Amer. J. 62, 1367–1377.

Six, J., Conant, R.T., Paul, E.A., Paustian, K., 2002. Stabilization mechanisms of soil organic matter: implications for C-saturation of soils. Plant Soil 241, 155–76.

Vanlauwe, B., Gachengo, C., Shepherd, K., Barrios, E., Cadisch, G., Palm, C., 2005a. Laboratory validation of a resource quality-based conceptual framework for organic matter management. Soil Sci. Soci. Amer. J. 69, 1135-1145.

Vanlauwe B., Aihou K., Tossah B.K., Diels J., Sanginga N., Merckx R. 2005b. *Senna siamea* trees recycle Ca from a Ca-rich subsoil and increase the topsoil pH in agroforestry systems in the West African derived savannah zone. Plant Soil 269, 285–296.

Van Noordwijk, M., Cadisch, G., Ong, C.K. (Eds.) 2004. Below-ground interactions in tropical agroecosystems: Concepts and models with multiple plant components. CAB International, Wallingford, UK.

Veen G.F., Wubs E.R.J., Bardgett R., Barrios E., Bradford M., Carvalho S., De Deyn G., de Vries F., Giller K., Kleijn D., Landis D., Rossing W.A.H., Schrama M., Six J., Struik P., van Gils S, Wiskerke H., van der Putten W.H., Vet L.E.M., 2019. Applying the aboveground-belowground interaction concept in agriculture: spatio-temporal scales matter. Front. Ecol. Evol. 7, 300.

Verchot, L.V., van Noordwijk, M., Kandji, S., Tomich, T., Ong, C., Albrecht, A., Mackensen, J., Bantilan, C., Anupama, K.V., Palm, C., 2007. Climate change: linking adaptation and mitigationthrough agroforestry. Mitig. Adapt. Strat. Global Change 12, 901–918.

Vitousek P.M., Porder S., Houlton B.Z., Chadwick O.A. 2010. Terrestrial phosphorus limitation: mechanisms, implications, and nitrogen–phosphorus interactions. Ecol. Applic. 20, 5–15.

Weil, R.R., Brady N.C., 2017. The nature and properties of soil. 15th Edition. Pearson, London.

Wong, M.T.F., Hairiah, K., Utami, R., Alegre, J., 2002. Managing acidity and aluminium toxicity in organic based agroecosystems. In: Ong C, van Noordwijk M, Cadisch G (eds). Belowground interactions in tropical agroecosystems with multiple plant components. CABI, Wallingford, UK, pp 143-156.