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Review

Enrichment Planting and Soil Amendments Enhance Carbon Sequestration and Reduce Greenhouse Gas Emissions in Agroforestry Systems: A Review

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Abstract: Agroforestry practices that intentionally integrate trees with crops and/or livestock in an agricultural production system could enhance carbon (C) sequestration and reduce greenhouse gas (GHG) emissions from terrestrial ecosystems, thereby mitigating global climate change. Beneficial management practices such as enrichment planting and the application of soil amendments can affect C sequestration and GHG emissions in agroforestry systems; however, such effects are not well understood. A literature review was conducted to synthesize information on the prospects for enhancing C sequestration and reducing GHG emissions through enrichment (i.e., in-fill) tree planting, a common practice in improving stand density within existing forests, and the application of organic amendments to soils. Our review indicates that in agroforests only a few studies have examined the effect of enrichment planting, which has been reported to increase C storage in plant biomass. The effect of adding organic amendments such as biochar, compost and manure to soil on enhancing C sequestration and reducing GHG emissions is well documented, but primarily in conventional crop production systems. Within croplands, application of biochar derived from various feedstocks, has been shown to increase soil organic C content, reduce CO_2 and N_2O emissions, and increase CH_4 uptake, as compared to no application of biochar. Depending on the feedstock used to produce biochar, biochar application can reduce N_2O emission by 3% to 84% as compared to no addition of biochars. On the other hand, application of compost emits less CO_2 and N_2O as compared to the application of manure, while the application of pelleted manure leads to more N_2O emission compared to the application of raw manure. In summary, enrichment planting and application of organic soil amendments such as compost and biochar will be better options than the application of raw manure for enhancing C sequestration and reducing GHG emissions. However, there is a shortage of data to support these practices in the field, and thus further research on the effect of these two areas of management intervention on C cycling will be imperative to developing best management practices to enhance C sequestration and minimize GHG emissions from agroforestry systems.

Keywords: climate change; manuring; manure pelleting; northern temperate; pyrolysis; information review

1. Introduction

Agriculture is the second largest emitter of greenhouse gases (GHG) after the energy sector, and is responsible for about 30% of global GHG emissions [1]. Agroforestry, the intentional integration of trees and/or shrubs with herbaceous crops and/or livestock in a production system, is a popular beneficial management practice (BMP) that can mitigate climate change by sequestering carbon (C) and reducing greenhouse gas (GHG) emissions [2–8]. The Intergovernmental Panel on Climate Change (IPCC) has recognized both afforestation and reforestation as important activities supporting C sequestration [9]. Agroforestry systems include many different permutations such as alley cropping, silvopasture, riparian buffers, savanna, forest farming, home-gardens, and woodlots, as well as other similar integrated land-use systems [10]. In all cases, agroforestry systems are recognized as a land use management framework that simultaneously integrates the dual goals of ecological conservation and socio-economic development [9–12].

The environmental service of sequestering C and reducing GHG emissions provided by agroforestry systems are relatively well-documented globally at various management systems as summarized in Section 3 below. However, data quantifying the specific role of management interventions to improve such benefits are rare, with only a few studies reporting on the potential benefits of enrichment planting [13,14] and organic amendment of soils [2,15]. Enrichment planting is commonly used for increasing the density of desired tree species in degraded (secondary) forests, particularly where these forests are low in density or occupied by less-desirable (i.e., non-productive) tree species. On the other hand, the addition of organic amendments to soils, including mulch, manure, or the application of other organic by-products from a feedlot or modified organic materials (such as biochar, composts, and manure pellets), is widely practiced in sole cropping systems but rare in agroforestry systems.

Manure is a widely available by-product from livestock production systems, particularly those involving the confined feeding of animals in large-scale livestock operations (poultry, swine, beef, dairy, etc.). Due to its high nutrient and C content, manure management and its application to soil plays a critical role in GHG emissions, including CH_4 and N_2O [16]. The Food and Agriculture Organization of the United Nations (FAO) estimated the CH_4 and N_2O emissions associated with manure storage and processing contribute 4.3% and 5.2% respectively, and N_2O emissions from the field applied and deposited manure contribute 16.4% of the GHG emissions in the global livestock supply chains [17]. Composting, despite emitting GHGs during storage, and dried pelleting, are two methods of conserving nutrients in manure and facilitating their slow release into the soil [18]. In the process, these methods reduce GHG emissions compared with raw manure application if applied to the field in an appropriate time such as avoiding wet conditions of soils [19]. Biochar is pyrolysed biomass consisting of around 50% or more recalcitrant organic material [20]. It is a promising soil amendment that improves physical and chemical properties of soils [21], as well as provides better environmental services such as improved nutrient cycling, increased C sequestration and reduce GHG emissions. However, studies on the effects of biochars in agroforestry systems are limited [22].

This paper reviews the current state of knowledge regarding opportunities to enhance C sequestration and reduce GHG emissions through two potential management interventions within agroforestry systems. The first is enrichment tree planting, and the second is the use of organic soil amendments. It starts with an overview of the impact of agroforestry in C sequestration and GHG emissions as documented by previous different empirical studies, and reviews reports at different temporal and spatial scales. Subsequent sections then review the role of enrichment planting and organic soil amendments and discuss the prospect of applying these interventions to agroforestry systems in order to enhance C sequestration and reduce GHG emissions. Finally, conclusions are drawn and areas of further research needs are identified on these two practices in order to further mitigate GHG emissions and promote climate change adaptation using agroforestry systems.

2. Methods of Literature Collection

A wide range of published literature was collected through searches using Google Scholar and ISI Web of Science with a Boolean defined by logical strings containing "and/or" with keywords "agroforestry", "environmental service", "enrichment planting", "greenhouse gas emission", "carbon sequestration", "biochar", "manure", "manure pellet", "composting", and "secondary forest".

More than 200 publications, both referred and non-reviewed, were found; they were further sorted with criteria of "carbon sequestration and agroforestry" and "greenhouse gas emission and agroforestry", "enrichment planting and secondary forest or agroforestry", "composting and greenhouse gas emission", "biochar and soil C sequestration", "raw manure and composted manure" and "manure pellet and soil carbon". With these criteria, the number of publications selected was reduced to 94. Among them, five publications were on enrichment planting, six on manure pelleting, 33 on manure, 10 on compost and management, and 40 on biochar. Key results found from these studies were compared focusing on C sequestration in vegetation and soils, as well as GHG emissions. A total of 82 publications closely related to the subject matter were used in this paper.

3. Role of Agroforestry in C Sequestration and Reducing Greenhouse Gas Emissions

Several studies have documented that the C sequestration potential of agroforestry systems varies depending on environment and specific management systems (Table 1). Estimated global C sequestration potential of agroforestry ranges between 12 and 228 Mg C ha^{-1}, leading to a net C sequestration potential of 1.1 to 2.2 Pg (1 Pg = 10^{15} g) over 50 years [23]. It is also estimated that improved management alone within existing agroforestry systems could sequester an additional 0.3 Mg C ha^{-1} y^{-1}, while undertaking land use changes from conventional cropland to agroforestry (crops combined with forests) could sequester an additional 3.1 Mg C ha^{-1} y^{-1} [24]. The SOC sequestration rate varies among agroforestry systems across different regions, ranging from 0.1 to 4.2 Mg C ha^{-1} y^{-1} depending upon the age of agroforests, and soil depths considered in the estimation [25]. In the early stage of practicing agroforestry, soil C can be lost from top soils; for example, a multi-strata agroforestry system in Ghana lost 0.4 Mg C ha^{-1} y^{-1} until 15 years after establishment, after which a small amount of SOC was stored (0.06 Mg C ha^{-1} y^{-1}) through 25 years age within the 0–15 cm soil layer (cited in [25]). Furthermore, the vegetation C sequestration rate differs by forest types. As an example, within windbreak systems, broadleaved trees demonstrated an almost double C storage capacity (4.39 ± 1.74 Mg C ha^{-1} y^{-1}) than conifer trees (2.45 ± 0.42 Mg C ha^{-1} y^{-1}) in a study involving nine ecoregions across the USA [26].

Table 1. Carbon sequestration rate, carbon stocks and greenhouse gas (GHG) emissions in different types of agroforestry systems. A negative flux shows consumption of GHGs.

Agroforestry/Management Activities/Location	Carbon/GHG Data	Reference
Above- and below-ground vegetation C sequestration rate (Mg C ha^{-1} y^{-1})		
Fodder bank, West Africa (7.5 years)	0.3	[9]
Tree-based inter-cropping, Canada (13 years)	0.8	
Agroforest, Western Oregon, USA (11 years)	1.1	
Agrisilviculture, India (5 years)	1.3	
Silvopasture, India (5 years)	6.6	
Home gardens, Togo (23 years)	4.3	
Shaded coffee, Togo (13 years)	6.3	
Home gardens, Indonesia (13 years)	8.0	
Cacao agroforest, Cameroon (26 years)	5.9	
Cacao Agroforest, Costa Rica (5 years)	10.3	
Cacao Agroforest, Costa Rica (10 years)	11.1	
Woodlots, Puerto Rico (4 years)	12.0	
Median C storage in different ecoregions		[27]
Semi-arid (5 years)	2.6	
Sub-humid (8 years)	6.1	
Humid (5 years)	10.0	
Temperate (30 years)	3.9	
Windbreak in U.S. ecoregions		[26]
Conifers	2.0–2.9	
Broadleaved	2.7–6.1	

Table 1. *Cont.*

Agroforestry/Management Activities/Location	Carbon/GHG Data	Reference
Shifting cultivation * in Peruvian Amazon and Indonesia	3.5	[28]
Improved fallow **		[4]
12-month-old fallow	5.3–13.2	
18-month-old fallow	17.4–31.9	
22-month-old fallow	21.3–30.5	
Vegetation C stock (Mg C ha^{-1})		
Improved fallow in Mediterranean	70	[28,29]
Potential C storage in six continents		
Africa, agrosilvicultural	29–53	
South America, agrosilvicultural	39–195	
Southeast Asia, agrosilvicultural	12–228	
Australia, silvopastoral	28–51	
North America, silvopastoral	90–198	
Northern Asia, silvopastoral	15–18	
Different agroforestry systems in sub-Saharan Africa:		[30]
Arid and semi-arid silvopastoral:		
pastoral/fruit	0.8–3.9	
pastoral/fuelwood	3.9–19.4	
pastoral/shelterbelt	1.7–1.8	
Humid silvopastoral:		
pastoral/fruit	2.0–8.6	
pastoral/fuelwood	5.1–24.7	
pastoral/shelterbelt	2.8–6.5	
Fruit/fuelwood	4.6–23.0	
Fruit/timber	33.3–71.3	
Fruit/shelterbelt	2.4–5.4	
Fuelwood/timber	36.4–86.8	
Fuelwood/shelterbelt	5.5–20.9	
Soil C sequestration rate (Mg C ha^{-1} y^{-1})		
Alley cropping, France (equivalent mass basis)		[31]
26–29 cm	0.25	
93–98 cm	0.35	
Improved fallow in Mediterranean	1.6	[28]
Soil C stock (Mg C ha^{-1})		
Three agroforestry systems, Alberta, Canada		[32]
0–10 cm		
Hedgerow (natural forest + crop)	77	
Shelterbelt (planted forest + crop)	67	
Silvopasture (natural forest + grassland)	101	
0–30 cm		
Hedgerow (natural forest + crop)	178	
Shelterbelt (planted forest + crop)	163	
Silvopasture (natural forest + grassland)	201	
Inter-cropping in sub-tropical China, 0–80 cm		[33]
Tree + shrub	93	
Tree + legume & cereal	79	
Tree + Oilseed & legume	74	
Humid tropics, 0–20 cm	25	[28]
Different agroforestry systems in Canada		[5]
Alley cropping, 0–40 cm (13–25 years)	71.1–125.4	
Alley cropping, 0–30 cm (8–9 years)	43.5–113.2	
Shelterbelt, 0–30 cm (various ages)	15–208	

Table 1. *Cont.*

Agroforestry/Management Activities/Location	Carbon/GHG Data	Reference
GHG emission rates (kg ha^{-1} y^{-1})		
Different agroforestry systems in Peruvian Amazon and Indonesia		
Shifting cultivation		
N_2O-N emission		
CH_4-C flux	0.8	
CO_2-C emission	−2.0	
Multi-strata agroforestry	5.9	
N_2O-N emission		[28]
CH_4-C flux	0.5	
CO_2-C emission	−2.0	
Peach-palm agroforestry	5.5	
N_2O-N emission		
CH_4-C flux	0.9	
CO_2-C emission	−1.5	
	5.8	
CO_2-C emission from agroforestry systems in Alberta		
Hedgerow (natural forest + crop)	16,425	
Shelterbelt (planted forest + crop)	10,950	[34]
Silvopasture (natural forest + grassland)	13,505	
CH_4-C emission		
Hedgerow (natural forest + crop)	−2.2	
Shelterbelt (planted forest + crop)	−1.8	
Silvopasture (natural forest + grassland)	−2.9	
Different agroforestry systems in Canada		
Alley Cropping		
CO_2-C emission	4900–6240	
Shelterbelts (combined with annual crops)		[5]
CO_2-C emission	1900–4000	
CH_4-C efflux	−0.15–−0.9	
N_2O-N efflux	0.25–3.0	
Shade coffee agroforestry, Sumatra		
N_2O-N	16	[4]
CH_4-C	−1.0	

* Plants are cut and burned to create the farming field—also called slash-and-burn system; ** rotation between cereal crops and tree-legume fallow.

Levels of C sequestration in vegetation and soils are known to vary among ecoregions. In the humid tropics, 70 and 25 Mg C ha^{-1} can be sequestered within vegetation and the top 20 cm of soil, respectively [28]. In Mediterranean regions, total C sequestration rates in vegetation and soils of different agroforestry systems can be up to 1.3 Mg C ha^{-1} y^{-1} [35]. In temperate climates, the potential C sequestration by aboveground vegetation of agroforestry systems could be as large as 2.1 × 10^9 Mg C y^{-1}, while in tropical regions, it could be 1.9 × 10^9 Mg C y^{-1} [36]. Collectively, these examples show that C sequestration rates and resulting C stocks vary widely, reflecting marked variation in climatic conditions, soil properties, vegetation types, and the ongoing management of agroforests. However, observed variation in estimates of C might also be due to the use of different methods for estimating soil C sequestration potential under contrasting conditions, coupled with the inherently high natural variability of soil C stocks within agroforestry systems associated with divergent agro-ecological zones [37].

Trees are also known to help reduce CH_4 and N_2O emissions, particularly in relation to neighboring cropland [5]. In the sub-tropics, agroforestry systems combining trees and inter-cropped shrubs store more C in vegetation and soils compared to systems with only trees or trees grown with legume or cereals as inter-cropped systems [38,39]. Similarly, windbreak and riparian forest buffers store significant amounts of C, in addition to providing other valuable ecosystem services such as improved water quality, biodiversity, and biomass feedstock availability [40]. Sequestered C in low-till croplands with adjacent treed windbreaks was 75% greater than in low-till lands without adjacent windbreaks [40]. Compared to sole herbland pastures, the presence of trees in the former leads to greater topsoil and subsoil C content, and larger litter inputs result in higher free and occluded organic matter (OM) fractions, and ultimately higher levels of stabilized SOM fractions [41]. A study in central Alberta, Canada showed that silvopastoral systems had higher SOC and lower GHG emissions compared to agroforestry systems containing either hedgerows or shelterbelts combined with annual cropland [32,34]. Mean SOC in the bulk soil at 0–10 cm depth was 81, 48 and 63 g kg^{-1} in the silvopasture, shelterbelt and hedgerow systems, respectively. Soil C in the more stable fine fraction (<53 μm) of the soil was higher in hedgerow systems (34 g kg^{-1}) compared to both shelterbelt and silvopasture systems (29 and 29 g kg^{-1}, respectively). Within each agroforestry system, total SOC and the SOC concentration within each size fraction was consistently greater in the forested land-use compared to the adjacent agricultural herbland [32]. The SOC stock in both the 0–10 cm and 10–30 cm soil layers were greater within the forested land cover type than in the adjacent herbland [42]. In terms of GHG emissions, the silvopasture system had 15% greater CH_4 uptake and 44% lower N_2O emission compared with the shelterbelt and silvopasture systems [34].

Despite their potential to mitigate GHG emissions, agroforestry systems can be a significant sink or source of GHGs depending upon management practices. In the humid tropics, agroforestry mitigated N_2O and CO_2 emissions from soils, and increased CH_4 uptake, compared to sole cropping systems [28]. While N_2O emission in the agroforestry system was as low as 3%, CO_2 emissions were 70% of the high input cropping systems, and CH_4 uptake was almost double that of the low input cropping system [33]. In contrast, management practices that disturbed soil and vegetation, such as tillage, burning of biomass, fertilization, and manuring, lead to net emissions of GHGs from soils and vegetation to the atmosphere [43]. Among different agroforestry systems, multi-strata systems reduce N_2O emissions and CH_4 oxidation, but emit similar CO_2 compared to shifting cultivation, crop/rubber agroforestry and short fallow systems [28].

4. Management Intervention to Enhance C Sequestration and Reduce GHG Emissions

4.1. Impacts of Enrichment Planting on C Sequestration and GHG Emissions

Enrichment planting, also known as in-fill or gap-planting, is commonly practiced to increase the density of desired tree species in degraded (secondary) forests [13,44], including those found in shelterbelts or silvopastoral plantations of agroforestry systems. Enrichment planting enables newly establishing trees to utilize available resources, including light, moisture and nutrients [45]. Agricultural systems that include trees generally store more C in deeper soil layers compared to treeless systems, and higher SOC content in the former has been associated with greater species richness of trees and tree density [4,46]. Enrichment planting in old fallow fields is beneficial in sequestering C, improving over-story tree diversity, and enhancing social, cultural, and ecosystem services [13]. The improved C storage observed after enrichment planting in eastern Panama was around 113 Mg C ha^{-1}, which is comparable to that in industrial teak plantations and primary forests [13].

The choice of tree species within plantations affects C storage in phytomass, necromass, and underlying soils. For example, after 40 years of growth in plantations, conifers had higher biomass and litter C, while broad-leaved forests had considerably more soil C [47]. The greater decomposition rate of broadleaf litter contributed favorably to soil C sequestration compared to that from conifer litter, but due to relatively steady photosynthetic rates throughout the year and high drought tolerance, conifers had more stable live biomass [47]. Understory vegetation biomass was also negatively correlated to tree-biomass, with conifer stands leading to less understory C mass due to increased canopy closure and associated light limitations [47].

Natural regeneration of vegetation in abandoned pasture land is known to sequester C in a manner similar to planted vegetation, although the rate of sequestration can be slower [48]. The aboveground C accumulation rate in 12–14 year-old forests was 5.6 Mg C ha^{-1} y^{-1}, and SOC accumulation rates were 1.49 Mg ha^{-1} y^{-1} [48]. These results indicate that natural regeneration of tree species can mimic enrichment planting after pasture abandonment.

4.2. Impact of Organic Soil Amendment on C Sequestration and Greenhouse Gas Emissions

Soil amendment with organic input is a common practice in conventional agricultural practices (e.g., on annual cropland or forage land), and involves manuring, mulching, green manuring and biochar addition. Agroforestry systems can provide various types of feedstock for bulking agents such as residues from annual crops, small woody biomass from pastures, leaf litter, as well as twigs, branches and woody biomass from trees for use in composting, pelleting or biochar production. In this section, we review relevant literature and summarize potential impacts of these amendments on C sequestration and GHG emissions in agroforestry.

4.2.1. Impacts of Biochar Applications

Biochar has a slow decomposition rate and its application to soils can sequester SOC compared to non-amended soils [49]. For example, 1.4 times higher total soil C was found in hardwood biochar amended soils compared to non-amended soils [49]. Biochar has been tested in different cropping systems to assess its impact on enhancing C sequestration and reducing GHG emissions. A review of a wide range of agro-ecosystems with biochar application showed that despite using a variety of feedstocks and crops, the resultant impact of biochar application was a decrease in GHG emissions by up to 66% in CO_2 and up to 50% in N_2O emissions [2]. In addition, biochar addition led to reduced leaching of plant nutrients and contamination of downstream water sources [2]. However, some biochar amended soils increased CO_2 emissions, which was attributed to increased soil porosity, lowered bulk density and higher pH, all of which may favor microorganism activity [2]. Biochar from wood and herbaceous feedstocks performed the best in reducing emissions (ca. −60%), while manure-based biochar was less effective, the latter of which altered N_2O emissions by −46% to +39% [50]. Biochar feedstock, pyrolysis conditions, and C/N ratios were key factors influencing the emissions of N_2O [50].

A previous meta-analysis of published data obtained from laboratory and field experiments to explore the effects of biochar on N_2O and CH_4 emissions reported nearly 50% less N_2O emissions across different soil types [51]. This same study indicated a potential to increase the uptake of CH_4 due to enhanced methanothrophy following biochar addition [51]. Effects of biochar on N_2O emissions also varied among feedstocks, with both woody and crop residue biochars decreasing emissions, while biochars derived from other feedstocks (e.g., manures, bio-solids, paper mill residues) had no significant effects [51]. A laboratory incubation study conducted across ten different soils in the USA and receiving the same hardwood biochar found no significant differences in the emissions of CO_2 and CH_4. However, this same study reported a decrease in N_2O emission (up to 63% less) across all soils after biochar applications [52]. Similarly, biochar produced from pine sawdust at 500 °C with or without steam activation decreased CO_2 and N_2O emission (up to 32% in forest soils), though no differences in CH_4 uptake were detected [53]. Pine biochar reduces GHG emissions by decreasing microbial and enzyme activities [53]. Moreover, by changing the physical (gas diffusivity, aggregation, water retention), chemical (e.g., pH, redox potential, availability of organic and mineral N and dissolved organic C, organo-mineral interactions), and biological properties (e.g., microbial community structure, microbial biomass and activity, macro faunal activity, N cycling enzymes) of soils, biochar influences N mineralization-immobilization, turnover, and nitrification or denitrification processes, all of which ultimately affect N_2O emissions [51].

The impact of biochar on GHG emissions within amended soils is dependent on both biochar and soil properties [20]. Relationships between the biochar and soil N dynamics revealed that adsorption of NH_4^+ and NO_3^- in biochar during the pyrolysis process decreased N loss during composting and after manure application, thus offering a mechanism for the slow release of fertilizer in the field [54]. Higher pyrolysis temperatures during the manufacture of biochar from manure and bio-solids also result in biochars with decreased hydrolysable organic N and increased aromatic N [54]. Short-term N_2O emissions are therefore likely to decrease following biochar application, though no clear information exists on the long-term effects of this practice. In summary, biochar input to agroecosystems represents a potential mitigation strategy for environmentally detrimental N losses, specifically as N_2O [54].

Biochar can also enhance the process of composting manure and reduce GHG emissions during composting and subsequent field applications. The impact of biochar addition in conjunction with composting, including their application to soils with manure and manure pellets, on GHG emissions and C sequestration, are summarized in Table 2. Application of biochar during the composting of chicken manure increased peak CO_2 emission, while emissions of both CH_4 and N_2O decreased [55,56]. Composting of cattle manure with added biochar increased aeration, and hence the activity of methanogens, which reduced CH_4 emission [15]. Biochar reduced N_2O and CH_4 emissions during field applications due to a change in the microenvironment for the microbial population, including soil water content, and availability of oxygen, N, and C [57]. In calcareous soils, biochar application alone increased total organic C stocks by 1.4 fold, while the application of biochar mixed with manure increased C levels by 1.7 fold [49].

Impacts of biochar on GHG emissions have shown mixed results depending on soil type, feedstock type and season of the application [58,59]. Biochar addition to upland soil increased CH_4 emissions by 37% during the summer, but had no effect in winter, while decreasing N_2O emissions up to 54% and 53% during the summer and winter seasons, respectively [58]. In Chernozemic soils amended with straw, and its biochar reduced N_2O emission but there were no significant effects on CH_4 or CO_2 emission compared with the unamended soils [59]. A soil-column experiment using non-treated soils and those amended with biochar prepared by pyrolysis of pig manure and spruce sawdust at 600 °C found no differences in N_2O, CH_4 and CO_2 emissions between the two treatments until they received an application of fresh pig manure during the 10th week [60]. After 10 weeks, cumulative GHG emissions were higher from soils amended with biochars and manure for up to four weeks compared to the non-treated soil. However, this same study found NO_3^-N leaching was 51% and 43% lower in pig manure biochar amended soils and wood biochar amended soils, respectively, compared to the pig manure only-amended soils [60].

Life cycle analysis is an emerging tool to link the full C footprint of products from their origin via different stages of the product supply chain [61–63]. A life cycle analysis of biochar systems comparing biochar produced from three feedstocks, namely corn stover, yard waste (waste from industrial-scale composting) and switchgrass energy crops, found that the net energy provided by corn stover and yard waste was negative (−864 and −885 kg CO_2e per Mg dry feedstock, respectively) while switchgrass was a net emitter (+36 kg CO_2e per Mg dry feedstock) [63]. These findings indicate that careful selection of biochar feedstock is required to avoid unintended environmental consequences, such as indirect increases in GHG emissions elsewhere in the global C and N cycles.

Table 2. Review of previous studies examining the effects of biochar and/or compost manure addition on relative changes in GHG emissions. Negative value shows reduction in GHG emissions.

Location	Experiment/Analysis	Variable	CO_2 (%)	CH_4 (%)	N_2O (%)	Reference	Remarks
Australia	Corn cropped red ferrosol amended with poultry litter (PL), PL biochar (PLB) and urea. Measured cumulative emissions for 57 days	PLB	1.3%	NA *	0.04%	[64]	Estimated CO_2-C and CH_4-C as percentage of TOC and N_2O-N as % of TN in 20 cm plough layer after biochar and manure addition.
		Raw PL	2.0%		0.27%		
		Urea	1.4%		0.16%		
		PLB + urea	1.6%		0.10%		
Ireland	Pig manure (PM) added to soil and further amended with biochar from pig manure (PMB) or spruce wood (WB). Cumulative emissions evaluated over 28 days 10 weeks after PM addition.	PM + PMB	2.3%	0.02%	3.8%	[60]	Estimated CO_2-C and CH_4-C as percentage of TOC and N_2O-N as % of TN in 20 cm plough layer after biochar and manure addition.
		PM + WB	2.1%	0.01%	4.1%		
		PM	5.5%	0.06%	2.1%		
Global	Meta-analysis on role of biochar from different feedstocks in regulating N_2O emissions in laboratory or field conditions. Feedstocks–biowaste [1] (BW), biosolids [2] (BS), manures or manure-based materials (MM), wood (W), herbaceous (H), lignocellulosic waste (LW).	Mean	NA	NA	−60 to 48%	[50]	[1] Biowaste = Municipal solid waste; [2] Biosolids = sewage sludge from water treatment plants.
		BW			−40%		
		BS			NS		
		MM			−46 to +39%		
		W			−60%		
		H			−60%		
		LW			−40%		
Global	Meta-analysis of published data on biochar application in soils from laboratory or field experiments.	Lab results	NA	NA	−60%	[51]	
		Field results			−40%		
		Mean			−50%		
USA	Fast pyrolysed (550 °C) Oak biochar (BC) applied to temperate soils from Colorado, Iowa, Michigan, and Minnesota, and 2 years incubation study for GHG emissions. Comparisons are to control treatments (lacking BC).	1% BC	8%	NA	−53.9%	[65]	
		5% BC	36%		−72.4%		
		10% BC	88%		−76.3%		
		20% BC	226%		−83.5%		
USA	Biochar produced at 550 °C from hardwood sawdust applied to soils from various locations.	Forest soils	120%	4.2%	−58.2%	[52]	% of change in GHG emissions after biochar addition compared to controlled soil with no biochar; Forest soils, $N = 2$; Agricultural soils, $N = 8$.
		Agricultural soils	75%	−0.9%	−54.4%		
Canada	Biochars produced at 300 and 550 °C with and without steam activation (BC-S) applied to forest and grassland soils at 1.5% mass basis. Comparisons are to control soils.	Forest soils				[53]	
		BC300	−0.1%	0.7%	−3.0%		
		BC300-S	4.2%	12.6%	−30.1%		
		BC500	−16.4%	18.1%	−27.5%		
		BC500-S	−5.7%	15.1%	−31.5%		
		Grassland soils					
		BC300	−2.7%	1.0%	−3.3%		

Table 2. *Cont.*

Location	Experiment/Analysis	Variable	CO_2 (%)	CH_4 (%)	N_2O (%)	Reference	Remarks
		BC300-S	−2.4%	−0.4%	−7.4%		
		BC500	−4.3%	4.3%	−14.8%		
		BC500-S	−2.2%	3.5%	−11.7%		
Germany	Manure compost from organic household wastes applied to 115-year old Norway Spruce plantation in silty and sandy soils at the rate of 6.3 kg m^{-2}.	Year 1 Silty soils Sandy soils Year 2 Silty soils Sandy soils	 24% 67% 20% 45%			[66]	% CO_2 emissions in fertilized area compared to control plots. Annual CO_2 emissions from control plots were 5.1 and 4.2 Mg C ha^{-1} y^{-1} in silty and sandy soils, respectively, in year 1, and 5.0 and 4.0 in year 2.
Brazil	Sewage sludge compost (SSC), sewage sludge (SS), mineral fertilizer (MF), and control (Ctrl) at the rate of 20 kg available N ha^{-1}.	SSC SS MF	90% 60% 13%	NS **	85% 37% 9%	[67]	% GHG emissions in fertilized area compared to control plots. CO_2 emission from control plots were 31.1 kg CO_2-C and 0.005 N_2O N (Mg ha^{-1}).

* NA = not available; ** NS = not significant.

Table 3. Summary of the previously documented effects of applying raw manure, composted manure, or manure pellets on subsequent C sequestration and GHG emissions in croplands and grasslands.

Location	Experiment	Treatment	CO_2 (kg ha^{-1})	CH_4 (kg ha^{-1})	N_2O (kg ha^{-1})	Reference	Remarks
Saskatchewan, Canada	Surface application, direct injection, and injection with soil aeration of swine effluent at 200 kg N ha^{-1} in no-till corn grain production.	Surface application Direct injection Combination with soil aeration	6900 8470 7370	1.2 2.6 2.1	7.3 4.7 6.9	[68]	Cumulative emissions for 141 days.
Quebec, Canada	Pig slurry applied to agricultural soils at 200 kg N ha^{-1} in spring and fall.	Fall Spring	997 1874	NA *	10.2 18.8	[69]	Seasonal cumulative measurement.
Quebec, Canada	Pig slurry (PS) applied for 19-year in loamy soil at 60 (PS60) or 120 (PS120) Mg ha^{-1} y^{-1}.	PS60 PS120	2820 6079	NA	4.9 13.1	[70,71]	12-month cumulative.
Germany	Soil amendments (50 mg N kg^{-1}) with cow manure (CM), poultry manure (PM), sheep and wheat straw compost (SWC), bio-waste compost (BWC) or calcium ammonium nitrate (CAN) in a laboratory experiment.	CM PM SWC BWC CAN Controlled	8118 2706 1804 6314 3608 1624	NA	0.4 0.4 0.1 0.1 0.1 0.0	[72]	

Table 3. *Cont.*

Location	Experiment	Treatment	CO_2 (kg ha^{-1})	CH_4 (kg ha^{-1})	N_2O (kg ha^{-1})	Reference	Remarks
Japan	Poultry manure (PM) and pelleted poultry (PP) manure application in Andisol at 120 kg N ha^{-1} in field and lab incubation experiment at two different water filled porosity (WFP) levels. Cumulative emission for 365 days.	Field condition PM PP Incubation—0.3 WFP Intact PP Ground PP Incubation—0.5 WFP Intact PP Ground PP	 NA 549 634 882 1060	NA	 1.3 5.0 0.9 4.4 10.1 67.8	[19]	Converted efflux to kg ha^{-1} at 5 cm soil depth (bulk density = 0.56 g cm^{-3})
USA	Incubation experiment with fine poultry manure (FPM) and pelleted poultry (PP) manure application in Cecil loamy sand at 55% and 90% of water filled porosity (WFP). N application rate was 307 kg N ha^{-1} equivalent.	55% WFP FPM PP 90% WFP FPM PP	 6584 6584 5267 4316		 53.3 65.8 1.6 15.7	[73]	Converted efflux to kg ha^{-1} at 15 cm soil depth (bulk density = 1.33 g cm^{-3}).
Scotland	Combination of dry pelleted and composted sewage sludge compared with liquid cattle slurry mixed with digested sewage sludge. Treatments were broadcasted sewage sludge pellet (DP): 15–17.5 t ha^{-1}, broadcasted compost sewage sludge (CP): 52–63.4 t ha^{-1}, injected digested liquid sewage sludge (LS) 60–120 t ha^{-1}, injected cattle slurry (CS) 5.9–10 t ha^{-1}.	Spring application DP CP LS CS Summer application DP CP LS CS	 10,633 11,367 11,367 13,200 18,700 22,000 20,900 22,000		 0.8 3.5 2.5 9.1 3.8 5.0 3.1 9.7	[18]	Manuring rate in grassland soils varied from year to year (kg N ha^{-1}); DP: 508–510, CP: 462–615, LS: 15–116, CS: 190–240.

* NA = not available.

4.2.2. Impacts of Raw Manure, Composted Manure, and Manure Pellets

Swine slurry (primarily liquid), farmyard manure (primarily from large mammals), and poultry manure are common by-products of livestock production that are recycled in the field as nutrient input to agricultural plants. Inorganic N content, labile C content and the water content in manure provide essential substrates to micro-organisms that affect GHG emissions from soil. However, GHGs can be produced and emitted to the atmosphere in each step from livestock confinement, to manure storage and treatment (i.e., handling and transport), and ultimately during application to the land [74]. Composting is a well-established manure management process because it utilizes livestock manure and residual biomass of livestock feed and bedding, and produces manure that has reduced pathogens and weed seeds [75]. On the other hand, manure pelleting, a physical method of densification, increases manure bulk density, reduces storage space requirements, reduces subsequent transportation costs, and makes these materials easier to handle. Cattle manure with 50% moisture content, and processed at a temperature of about 40 °C and a pressure of 6 MPa, resulted in maximum pellet durability [76].

In the field, slurry and manure application methods and their forms affect GHG emissions [68,77]. Effects of raw farmyard manure, compost and pelleted manure application on soil C sequestration and GHG emissions are summarized in Table 3. Conventional injected pig slurry emitted greater CH_4 compared to injection with soil aeration, while manure spread on the surface emitted higher N_2O than both types of injection [68]. Slurry application season, method and rate all affected CO_2 and N_2O emissions in the field [69–71].

The inclusion of compost into soil provides better nutrient input compared to raw manure from the perspective of C sequestration [77]. For example, four years after application about 36% of applied compost remained in the soil as sequestered C, as compared to only 25% of applied raw manure [77]. Composting increases the aromatic bonds and reduces the soluble C/N ratio in manure compost [75], which leads to the slow release of nutrients. Composted manures are more effective in reducing N_2O emissions than raw manures for soil amendments [73]. In general, N_2O is produced through the denitrification process of organic fertilizers [78] while nitrification is the most important process for inorganic fertilizer. From one to five percent of total N applied from organic manure was emitted with emission rates depending largely on soil nitrate levels, dissolved organic carbon (DOC) content and aeration, as well as soil temperature, moisture and pH [79].

Effects of manure from different livestock and composts also vary in GHG emissions from soils. Raw cattle manure emitted higher amounts of CO_2 (~1 g kg^{-1} dry matter), followed by bio-waste composts, poultry manure, and sheep waste compost, within arable soils in Germany [72]. Application of cattle manure and straw mixed together as a compost enhanced C sequestration and reduced N_2O emissions; thus, composting of manure containing high lignin, such as rice-husk or wheat straw, is beneficial [72]. Such compost reduces soil pH, which slows down the nitrification process and reduces N_2O emissions [72].

Pellets derived from a mix of manure and urea enhanced nutrient use efficiency via the slow-release of nutrients and led to increased crop yields [80]. Unlike composting, pelleted manures are less effective in reducing GHG emissions than raw (i.e., untreated) manure [19]. Annual cumulative emission of N_2O from pelleted poultry manure applied in the field was almost four times higher than that from raw poultry manure. Similarly, higher CO_2 emissions were detected from soils amended with intact pelleted poultry manure compared to the application of ground pelleted poultry manure under anaerobic incubation [19]. N_2O emission was 154 mg N kg^{-1} dry soils from intact pelleted manure-amended soils, which was almost seven times higher than from ground pelleted manure-amended soil [19]. Soils emitted significantly higher N_2O when treated with pelletized poultry litter (6.8% of applied N) than for fine-particle litter (5.5%) at 55% of water field capacity (WFP). In contrast, at 90% of WFP, fine-particle litter treated soils emitted higher N_2O (3.4%) than soils receiving pelletized litter (1.5%), indicating GHG responses to pellet application depended on moisture, with pellets leading to more GHG if moisture is low [73]. Reported CO_2 emissions ranged from 29 to

43 g C kg^{-1} across moisture levels, though they were not statistically different. Results indicate that N_2O emissions, but not CO_2 emissions, from soils treated with poultry litter depend on its physical characteristics of litter and soil water regime. Diminishing rates of N_2O emission after the application of manure pellets to soil are attributed to the polymer chain reaction defined by the specific type of nitrite reductase encoded by the *nirS* gene, which fluctuated with time; however, the *nirK* gene remains relatively stable, making *nirS* responsible for the denitrification process of N in manure pellets [15].

In forest ecosystems, application of organic and inorganic fertilizer has shown different effects on GHG emissions depending upon geographic location. In Germany the application of composted household waste manure increased CO_2 emission by 24% in silty soils and by 66% in sandy soils compared to control plots [66]. On the other hand the application of organic and inorganic fertilizer to tropical forest in Brazil increased CO_2 emission by 90%, and 60% in composted sewage sludge, and raw sewage sludge amended sites, respectively, compared to emissions from controlled plots [67]. Surprisingly, N_2O emissions were 85 and 37 fold higher in the composted sewage sludge amended and raw sewage sludge amended plots compared to control plots [67].

5. Conclusions

Agroforestry has emerged as a holistic land use practice creating a win-win scenario for environment and society [81]. Combining woody vegetation with cropping and livestock production via agroforestry systems increases total production, enhances food and nutrition security and mitigates the effects of climate change [81,82]. Carbon sequestration and GHG emissions in agroforestry systems are complex and depend on various biophysical factors such as climatic conditions, soil properties, water regime, vegetation characteristics, and the site-specific management practices undertaken, including inputs. The ability of an agroforestry system to enhance C sequestration and reduce GHG emissions depends on the region-specific biophysical condition. Several estimates showed that agroforestry systems in temperate regions have higher C pools than other climatic regions [28,36]. Silvopastoral systems were found to be superior in terms of both C sequestration and reducing GHG emissions [32,34,41] compared to agroforestry systems that included annual cropland. Interventions like enrichment planting and organic amendment of soils to slow down nutrient release are also site-specific in regulating their effectiveness [13]. Our review indicates that broadleaved tree species used in enrichment planting contribute towards more soil C, while conifers sequester more in their biomass over the long-term [47]. Results of this literature review showed that the effects of enrichment planting in agroforestry are not studied widely. The paucity of literature on this topic limits the drawing of conclusions with respect to the type of enrichment planting that will be most effective in optimizing ecosystem goods and services from agroforestry.

Livestock manure applied to soils in the form of pellets, compost or biochar, can play a significant role in increasing C sequestration and reducing GHG emissions. Soil amendment with biochar increases soil porosity, and aids water and nutrient retention, thereby creating a favorable situation for nutrient uptake by plants. By enhancing biomass production, biochar can play an important role in sequestering C in vegetation and soils. Additionally, decreased emissions of N_2O and CO_2, and increased uptake of CH_4, have been reported in the literature after biochar application. However, many of these studies were carried out in annual cropping systems, leaving a substantial knowledge gap with respect to their effectiveness in agroforestry systems. Further studies on the specific effects of organic amendments to soils within either the treed area or adjacent cropland may provide a better idea on how agroforestry systems can be collectively managed to achieve greater C sequestration and reduce GHG emissions. In general, raw manure management and field applications of manure were found to be sources of CO_2, CH_4, and N_2O emissions, although the magnitude of these GHG emissions varied with application season, methods and amounts. Composting and pelleting of manures can reduce GHG emissions while making manure more convenient to store and use. A more thorough study is warranted to better understand the relationship between different types of feedstocks and their capacity to enhance C sequestration and reduce GHG emissions within agroforestry systems.

Overall, this review found that enhancement of C sequestration and reducing GHG emissions in agroforestry are possible through management interventions. Enrichment planting practiced in secondary forestry management and organic amendment of soils in conventional cropping systems should be further explored within an agroforestry management framework, as potential interventions to enhance C sequestration and reduce GHG emissions. Further studies will provide better evidence of such beneficial practices for environment, economy, and society.

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References

1. McIntyre, B.D.; Herren, H.R.; Wakhungu, J.; Watson, R.T. *International Assessment of Agricultural Knowledge, Science and Technology for Development (IAASTD): Global Report*; Island Press, The Center for Resource Economics: Washington, DC, USA, 2009.
2. Stavi, I.; Lal, R. Agroforestry and biochar to offset climate change: A review. *Agron. Sustain. Dev.* **2013**, *33*, 81–96. [CrossRef]
3. Schoeneberger, M.; Bentrup, G.; de Gooijer, H.; Soolanayakanahally, R.; Sauer, T.; Brandle, J.; Zhou, X.; Current, D. Branching out: Agroforestry as a climate change mitigation and adaptation tool for agriculture. *J. Soil Water Conserv.* **2012**, *67*, 128A–136A. [CrossRef]
4. Verchot, L.V.; Van Noordwijk, M.; Kandji, S.; Tomich, T.; Ong, C.; Albrecht, A.; Mackensen, J.; Bantilan, C.; Anupama, K.V.; Palm, C. Climate change: Linking adaptation and mitigation through agroforestry. *Mitig. Adapt. Strateg. Glob. Chang.* **2007**, *12*, 901–918. [CrossRef]
5. Baah-Acheamfour, M.; Chang, S.X.; Bork, E.W.; Carlyle, C.N. The potential of agroforestry to reduce atmospheric greenhouse gases in Canada: Insight from pairwise comparisons with traditional agriculture, data gaps and future research. *For. Chron.* **2017**, *93*, 180–189. [CrossRef]
6. Jose, S. Agroforestry for ecosystem services and environmental benefits: An overview. *Agrofor. Syst.* **2009**, *76*, 1–10. [CrossRef]
7. Mbow, C.; Smith, P.; Skole, D.; Duguma, L.; Bustamante, M. Achieving mitigation and adaptation to climate change through sustainable agroforestry practices in Africa. *Curr. Opin. Environ. Sustain.* **2014**, *6*, 8–14. [CrossRef]
8. Newaj, R.; Chaturvedi, O.P.; Handa, A.K. Recent development in agroforestry research and its role in climate change adaptation and mitigation. *Indian J. Agrofor.* **2016**, *18*, 1–9.
9. Nair, P.K.R.; Nair, V.D.; Kumar, B.M.; Showalter, J.M. Carbon sequestration in agroforestry systems. *Adv. Agron.* **2010**, *108*, 237–307.
10. Kumar, B.M.; Nair, P.K.R. *Carbon Sequestration Potential of Agroforestry Systems: Opportunities and Challenges*; Springer: Berlin, Germany, 2011; Volume 8.
11. Montagnini, F.; Nair, P.K.R. Carbon sequestration: An underexploited environmental benefit of agroforestry systems. *Agrofor. Syst.* **2004**, *61*, 281.
12. Nuberg, I.; Reid, R.; George, B. *Agroforestry as Integrated Natural Resource Management*; CSIRO Publishing: Collingwood, Australia, 2009.
13. Paquette, A.; Hawryshyn, J.; Senikas, A.; Potvin, C. Enrichment planting in secondary forests: A promising clean development mechanism to increase terrestrial carbon sinks. *Ecol. Soc.* **2009**, *14*, 31. [CrossRef]

14. Suryanto, P.; Putra, E.T.S. Traditional enrichment planting in agroforestry marginal land Gunung Kidul, Java, Indonesia. *J. Sustain. Dev.* **2012**, *5*, 77. [CrossRef]
15. Sonoki, T.; Furukawa, T.; Mizumoto, H.; Jindo, K.; Aoyama, M.; Sanchez-Monedero, M.A. Impacts of biochar addition on methane and carbon dioxide emissions during composting of cattle manure. In Proceedings of the Asia Pacific Biochar Conference, Kyoto, Japan, 15–18 September 2011.
16. Zhongqi, H.E.; Pagliari, P.H.; Waldrip, H.M. Applied and Environmental Chemistry of Animal Manure: A Review. *Pedosphere* **2016**, *26*, 779–816.
17. Gerber, P.J.; Steinfeld, H.; Henderson, B.; Mottet, A.; Opio, C.; Dijkman, J.; Falcucci, A.; Tempio, G. *Tackling Climate Change Through Livestock—A Global Assessment of Emissions and Mitigation Opportunities*; Food and Agriculture Organization of the United Nations (FAO): Rome, Italy, 2013.
18. Ball, B.C.; McTaggart, I.P.; Scott, A. Mitigation of greenhouse gas emissions from soil under silage production by use of organic manures or slow-release fertilizer. *Soil Use Manag.* **2004**, *20*, 287–295. [CrossRef]
19. Hayakawa, A.; Akiyama, H.; Sudo, S.; Yagi, K. N_2O and NO emissions from an Andisol field as influenced by pelleted poultry manure. *Soil Biol. Biochem.* **2009**, *41*, 521–529. [CrossRef]
20. Spokas, K.A.; Reicosky, D.C. Impacts of sixteen different biochars on soil greenhouse gas production. *Ann. Environ. Sci.* **2009**, *3*, 4.
21. Hansen, V.; Müller-Stöver, D.; Munkholm, L.J.; Peltre, C.; Hauggaard-Nielsen, H.; Jensen, L.S. The effect of straw and wood gasification biochar on carbon sequestration, selected soil fertility indicators and functional groups in soil: An incubation study. *Geoderma* **2016**, *269*, 99–107. [CrossRef]
22. Stavi, I. Biochar use in forestry and tree-based agro-ecosystems for increasing climate change mitigation and adaptation. *Int. J. Sustain. Dev. World Ecol.* **2013**, *20*, 166–181. [CrossRef]
23. Albrecht, A.; Kandji, S.T. Carbon sequestration in tropical agroforestry systems. *Agric. Ecosyst. Environ.* **2003**, *99*, 15–27. [CrossRef]
24. Watson, R.T.; Noble, I.R.; Bolin, B.; Ravindranath, N.H.; Verardo, D.J.; Dokken, D.J. *Land Use, Land-Use Change and Forestry. A Special Report of the Intergovernmental Panel on Climate Change (IPCC)*; Cambridge University: Cambridge, UK, 2000.
25. Lorenz, K.; Lal, R. Soil organic carbon sequestration in agroforestry systems. A review. *Agron. Sustain. Dev.* **2014**, *34*, 443–454. [CrossRef]
26. Possu, W.B.; Brandle, J.R.; Domke, G.M.; Schoeneberger, M.; Blankenship, E. Estimating carbon storage in windbreak trees on U.S. agricultural lands. *Agrofor. Syst.* **2016**, *90*, 889–904. [CrossRef]
27. Schroeder, P. Agroforestry systems: Integrated land use to store and conserve carbon. *Clim. Res.* **1993**, *3*, 53–60. [CrossRef]
28. Mutuo, P.K.; Cadisch, G.; Albrecht, A.; Palm, C.A.; Verchot, L. Potential of agroforestry for carbon sequestration and mitigation of greenhouse gas emissions from soils in the tropics. *Nutr. Cycl. Agroecosyst.* **2005**, *71*, 43–54. [CrossRef]
29. Kandji, S.T.; Verchot, L.V.; Mackensen, J.; Boye, A.; Van Noordwijk, M.; Tomich, T.P.; Ong, C.K.; Albrecht, A.; Palm, C.A.; Garrity, D.P. Opportunities for linking climate change adaptation and mitigation through agroforestry systems. In *World Agroforestry into the Future*; World Agroforestry Centre, ICRAF: Nairobi, Kenya, 2006; pp. 113–121.
30. Unruh, J.; Houghton, R.; Lefebvre, P. Carbon storage in agroforestry: An estimate for sub-Saharan Africa. *Clim. Res.* **1993**, *3*, 39–52. [CrossRef]
31. Cardinael, R.; Chevallier, T.; Barthès, B.G.; Saby, N.P.A.; Parent, T.; Dupraz, C.; Bernoux, M.; Chenu, C. Impact of alley cropping agroforestry on stocks, forms and spatial distribution of soil organic carbon—A case study in a Mediterranean context. *Geoderma* **2015**, *259*, 288–299. [CrossRef]
32. Baah-Acheamfour, M.; Carlyle, C.N.; Bork, E.W.; Chang, S.X. Trees increase soil carbon and its stability in three agroforestry systems in central Alberta, Canada. *For. Ecol. Manag.* **2014**, *328*, 131–139. [CrossRef]
33. Wang, G.; Welham, C.; Feng, C.; Chen, L.; Cao, F. Enhanced Soil Carbon Storage under Agroforestry and Afforestation in Subtropical China. *Forests* **2015**, *6*, 2307–2323. [CrossRef]
34. Baah-Acheamfour, M.; Carlyle, C.N.; Lim, S.-S.; Bork, E.W.; Chang, S.X. Forest and grassland cover types reduce net greenhouse gas emissions from agricultural soils. *Sci. Total Environ.* **2016**, *571*, 1115–1127. [CrossRef] [PubMed]

35. Chenu, C.; Cardinael, R.; Chevallier, T.; Germon, A.; Jourdan, C.; Dupraz, C.; Barthès, B.; Bernoux, M. The contribution of agroforestry systems to climate change mitigation—Assessment of C storage in soils in a Mediterranean context. In *Our Common Future under Climate Change*; CFCC: Paris, France, 2015.
36. Oelbermann, M.; Voroney, R.P.; Gordon, A.M. Carbon sequestration in tropical and temperate agroforestry systems: A review with examples from Costa Rica and southern Canada. *Agric. Ecosyst. Environ.* **2004**, *104*, 359–377. [CrossRef]
37. Atangana, A.; Khasa, D.; Chang, S.; Degrande, A. Carbon Sequestration in Agroforestry Systems. In *Tropical Agroforestry*; Springer: Dordrecht, The Netherlands, 2014; pp. 217–225.
38. He, Y.; Qin, L.; Li, Z.; Liang, X.; Shao, M.; Tan, L. Carbon storage capacity of monoculture and mixed-species plantations in subtropical China. *For. Ecol. Manag.* **2013**, *295*, 193–198. [CrossRef]
39. Paquette, A.; Bouchard, A.; Cogliastro, A. Survival and growth of under-planted trees: A meta-analysis across four biomes. *Ecol. Appl.* **2006**, *16*, 1575–1589. [CrossRef]
40. Schoeneberger, M.M. Agroforestry: Working trees for sequestering carbon on agricultural lands. *Agrofor. Syst.* **2009**, *75*, 27–37. [CrossRef]
41. Hoosbeek, M.R.; Remme, R.P.; Rusch, G.M. Trees enhance soil carbon sequestration and nutrient cycling in a silvopastoral system in south-western Nicaragua. *Agrofor. Syst.* **2016**, *92*, 263–273. [CrossRef]
42. Baah-Acheamfour, M.; Chang, S.X.; Carlyle, C.N.; Bork, E.W. Carbon pool size and stability are affected by trees and grassland cover types within agroforestry systems of western Canada. *Agric. Ecosyst. Environ.* **2015**, *213*, 105–113. [CrossRef]
43. Dixon, R.K. Agroforestry systems: Sources of sinks of greenhouse gases? *Agrofor. Syst.* **1995**, *31*, 99–116. [CrossRef]
44. Montagnini, F.; Eibl, B.; Grance, L.; Maiocco, D.; Nozzi, D. Enrichment planting in overexploited subtropical forests of the Paranaense region of Misiones, Argentina. *For. Ecol. Manag.* **1997**, *99*, 237–246. [CrossRef]
45. McGuire, J.P.; Mitchell, R.J.; Moser, E.B.; Pecot, S.D.; Gjerstad, D.H.; Hedman, C.W. Gaps in a gappy forest: Plant resources, longleaf pine regeneration, and understory response to tree removal in longleaf pine savannas. *Can. J. For. Res.* **2001**, *31*, 765–778. [CrossRef]
46. Nair, P.K.R.; Nair, V.D.; Kumar, B.M.; Haile, S.G. Soil carbon sequestration in tropical agroforestry systems: A feasibility appraisal. *Environ. Sci. Policy* **2009**, *12*, 1099–1111. [CrossRef]
47. Gao, Y.; Cheng, J.; Ma, Z.; Zhao, Y.; Su, J. Carbon storage in biomass, litter, and soil of different plantations in a semiarid temperate region of northwest China. *Ann. For. Sci.* **2014**, *71*, 427–435. [CrossRef]
48. Feldpausch, T.R.; Rondon, M.A.; Fernandes, E.; Riha, S.J.; Wandelli, E. Carbon and nutrient accumulation in secondary forests regenerating on pastures in central Amazonia. *Ecol. Appl.* **2004**, *14*, 164–176. [CrossRef]
49. Lentz, R.D.; Ippolito, J.A. Biochar and manure affect calcareous soil and corn silage nutrient concentrations and uptake. *J. Environ. Qual.* **2012**, *41*, 1033–1043. [CrossRef] [PubMed]
50. Cayuela, M.L.; Van Zwieten, L.; Singh, B.P.; Jeffery, S.; Roig, A.; Sánchez-Monedero, M.A. Biochar's role in mitigating soil nitrous oxide emissions: A review and meta-analysis. *Agric. Ecosyst. Environ.* **2014**, *191*, 5–16. [CrossRef]
51. Van Zwieten, L.; Kammann, C.; Cayuela, M.; Singh, B.P.; Joseph, S.; Kimber, S.; Donne, S.; Clough, T.; Spokas, K.A. Biochar effects on nitrous oxide and methane emissions from soil. In *Biochar for Environmental Management: Science, Technology and Implementation*; Routledge: New York, NY, USA, 2015.
52. Thomazini, A.; Spokas, K.; Hall, K.; Ippolito, J.; Lentz, R.; Novak, J. GHG impacts of biochar: Predictability for the same biochar. *Agric. Ecosyst. Environ.* **2015**, *207*, 183–191. [CrossRef]
53. Pokharel, P.; Kwak, J.-H.; Ok, Y.S.; Chang, S.X. Pine sawdust biochar reduces GHG emission by decreasing microbial and enzyme activities in forest and grassland soils in a laboratory experiment. *Sci. Total Envirn.* **2018**, *625*, 1247–1256. [CrossRef]
54. Clough, T.J.; Condron, L.M.; Kammann, C.; Müller, C. A review of biochar and soil nitrogen dynamics. *Agronomy* **2013**, *3*, 275–293. [CrossRef]
55. Jia, X.; Wang, M.; Yuan, W.; Shah, S.; Shi, W.; Meng, X.; Ju, X.; Yang, B. N_2O emission and nitrogen transformation in chicken manure and biochar co-composting. *Trans. Am. Soc. Agric. Biol. Eng.* **2016**, *59*, 1277–1283.
56. Jia, X.; Wang, M.; Yuan, W.; Ju, X.; Yang, B. The influence of biochar addition on chicken manure composting and associated methane and carbon dioxide emissions. *BioResources* **2016**, *11*, 5255–5264. [CrossRef]

57. Lehmann, J.; Rillig, M.C.; Thies, J.; Masiello, C.A.; Hockaday, W.C.; Crowley, D. Biochar effects on soil biota—A review. *Soil Biol. Biochem.* **2011**, *43*, 1812–1836. [CrossRef]
58. Wang, J.; Pan, X.; Liu, Y.; Zhang, X.; Xiong, Z. Effects of biochar amendment in two soils on greenhouse gas emissions and crop production. *Plant Soil* **2012**, *360*, 287–298. [CrossRef]
59. Wu, F.; Jia, Z.; Wang, S.; Chang, S.X.; Startsev, A. Contrasting effects of wheat straw and its biochar on greenhouse gas emissions and enzyme activities in a Chernozemic soil. *Biol. Fertil. Soils* **2013**, *49*, 555–565. [CrossRef]
60. Troy, S.M.; Lawlor, P.G.; O'Flynn, C.J.; Healy, M.G. Impact of biochar addition to soil on greenhouse gas emissions following pig manure application. *Soil Biol. Biochem.* **2013**, *60*, 173–181. [CrossRef]
61. Homagain, K.; Shahi, C.; Luckai, N.; Sharma, M. Life cycle cost and economic assessment of biochar-based bioenergy production and biochar land application in Northwestern Ontario, Canada. *For. Ecosyst.* **2016**, *3*, 21. [CrossRef]
62. Homagain, K.; Shahi, C.; Luckai, N.; Sharma, M. Life cycle environmental impact assessment of biochar-based bioenergy production and utilization in Northwestern Ontario, Canada. *J. For. Res.* **2015**, *26*, 799–809. [CrossRef]
63. Roberts, K.G.; Gloy, B.A.; Joseph, S.; Scott, N.R.; Lehmann, J. Life cycle assessment of biochar systems: Estimating the energetic, economic, and climate change potential. *Environ. Sci. Technol.* **2009**, *44*, 827–833. [CrossRef] [PubMed]
64. Van Zwieten, L.; Kimber, S.W.L.; Morris, S.G.; Singh, B.P.; Grace, P.R.; Scheer, C.; Rust, J.; Downie, A.E.; Cowie, A.L. Pyrolysing poultry litter reduces N_2O and CO_2 fluxes. *Sci. Total Environ.* **2013**, *465*, 279–287. [CrossRef] [PubMed]
65. Stewart, C.E.; Zheng, J.; Botte, J.; Cotrufo, M.F. Co-generated fast pyrolysis biochar mitigates green-house gas emissions and increases carbon sequestration in temperate soils. *GCB Bioenergy* **2013**, *5*, 153–164. [CrossRef]
66. Borken, W.; Muhs, A.; Beese, F. Application of compost in spruce forests: Effects on soil respiration, basal respiration and microbial biomass. *For. Ecol. Manag.* **2002**, *159*, 49–58. [CrossRef]
67. De Urzedo, D.I.; Franco, M.P.; Pitombo, L.M.; do Carmo, J.B. Effects of organic and inorganic fertilizers on greenhouse gas (GHG) emissions in tropical forestry. *For. Ecol. Manag.* **2013**, *310*, 37–44. [CrossRef]
68. Sistani, K.R.; Warren, J.G.; Lovanh, N. Quantification of Greenhouse Gas Emissions from Soil Applied Swine Effluent by Different Methods. In *Livestock Environment VIII, 31 August–4 September 2008, Iguassu Falls, Brazil*; American Society of Agricultural and Biological Engineers: St. Joseph, MI, USA, 2008; p. 26.
69. Rochette, P.; Angers, D.A.; Chantigny, M.H.; Bertrand, N.; Côté, D. Carbon dioxide and nitrous oxide emissions following fall and spring applications of pig slurry to an agricultural soil. *Soil Sci. Soc. Am. J.* **2004**, *68*, 1410–1420. [CrossRef]
70. Rochette, P.; van Bochove, E.; Prévost, D.; Angers, D.A.; Bertrand, N. Soil carbon and nitrogen dynamics following application of pig slurry for the 19th consecutive year II. Nitrous oxide fluxes and mineral nitrogen. *Soil Sci. Soc. Am. J.* **2000**, *64*, 1396–1403. [CrossRef]
71. Rochette, P.; Angers, D.A. Soil carbon and nitrogen dynamics following application of pig slurry for the 19th consecutive year I. Carbon dioxide fluxes and microbial biomass carbon. *Soil Sci. Soc. Am. J.* **2000**, *64*, 1389–1395. [CrossRef]
72. Shah, A.; Lamers, M.; Streck, T. N_2O and CO_2 emissions from South German arable soil after amendment of manures and composts. *Environ. Earth Sci.* **2016**, *75*, 1–12. [CrossRef]
73. Cabrera, M.L.; Chiang, S.C.; Merka, W.C.; Pancorbo, O.C.; Thompson, S.A. Nitrous oxide and carbon dioxide emissions from pelletized and nonpelletized poultry litter incorporated into soil. *Plant Soil* **1994**, *163*, 189–195. [CrossRef]
74. Chadwick, D.; Sommer, S.; Thorman, R.; Fangueiro, D.; Cardenas, L.; Amon, B.; Misselbrook, T. Manure management: Implications for greenhouse gas emissions. *Anim. Feed Sci. Technol.* **2011**, *166*, 514–531. [CrossRef]
75. Huang, G.F.; Wu, Q.T.; Wong, J.W.C.; Nagar, B.B. Transformation of organic matter during co-composting of pig manure with sawdust. *Bioresour. Technol.* **2006**, *97*, 1834–1842. [CrossRef] [PubMed]
76. Zafari, A.; Kianmehr, M.H. Effect of temperature, pressure and moisture content on durability of cattle manure pellet in open-end die method. *J. Agric. Sci.* **2012**, *4*, 203. [CrossRef]
77. Eghball, B.; Ginting, D. *Carbon Sequestration Following Beef Cattle Feedlot Manure, Compost, and Fertilizer Applications*; Nebraska Beef Cattle Reports; University of Nebraska: Lincoln, NE, USA, 2003.

78. Yamane, T. Denitrifying bacterial community in manure compost pellets applied to an Andosol upland field. *Soil Sci. Plant Nutr.* **2013**, *59*, 572–579. [CrossRef]
79. Biala, J.; Rowlings, D.W.; De Rosa, D.; Grace, P. Effects of using raw and composted manures on nitrous oxide emissions: A review. In Proceedings of the XXIX International Horticultural Congress on Horticulture: Sustaining Lives, Livelihoods and Landscapes (IHC2014), Brisbane, Australia, 17–22 August 2014; pp. 425–430.
80. Alemi, H.; Kianmehr, M.; Borghaee, A. Effect of pellet processing of fertilizer on slow-release nitrogen in soil. *Asian J. Plant Sci.* **2010**, *9*, 74. [CrossRef]
81. Nair, P.K.R. The coming of age of agroforestry. *J. Sci. Food Agric.* **2007**, *87*, 1613–1619. [CrossRef]
82. Nair, P.K.R. Agroforestry Systems and Environmental Quality. *J. Environ. Qual.* **2011**, *40*, 784–790. [CrossRef] [PubMed]