Carbon Footprints

Review

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Carbon sequestration by forests and agroforests: a reality check for the United States

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Abstract

Climate change is a major global threat affecting food security and sustainability. Land use systems involving trees have the potential to positively impact climate change by reducing atmospheric carbon dioxide (CO_2) and providing long-term carbon (C) storage. This review evaluated the C sequestration potential of two major land use systems of the United States (US) involving trees, forests and agroforests, which can also provide other ecosystem services.The estimated total forest C stock on forest land in the US in 1990 was 50,913 Tg and another 1885 Tg remained in harvested wood and discarded wood products. From 1990 to 1995, total C stock rose by 2%, and from 2000 to 2005, it rose by 1.7%. The US forests collectively lose (flux) about 200 Mg C y^{-1} from disturbance and harvesting. Currently, about 12% of the conterminous US forest land is at high or very high risk of wildfire. Annually, insects and diseases could transfer ~ 21 Tg of live aboveground biomass to litter and woody debris pools. A scenario that targets an afforestation policy for rural landowners in the eastern US and a reforestation policy targeting understocked federal forest lands in the western would improve US annual sequestration compared to the baseline of 323 Tg CO_2 eq yr^{-1} in 2015 to 469 Tg CO_2 eq yr^{-1} in 2050.Agroforestry offers greater potential to increase C sequestration of predominantly agriculture-dominated landscapes than monocrop agriculture by storing C in above- and belowground biomass, soil, and living and dead organisms and further extending the duration of C in soils. The estimated total C sequestration of current alley cropping (211,938 ha), riparian buffers (640,732 ha), silvopasture (34 Mha), and windbreak (2.37 Mha) practices is 219 Tg C yr^{-1}. The total C sequestration would be 240 Tg C yr^{-1} with 5% of the US cropland converted to alley cropping (3.7 Tg yr^{-1}), 15-m wide riparian buffers on both sides of 5% of the total stream length (4.75 Tg yr^{-1}), 34 Mha converted to silvopasture (207 Tg yr^{-1}), and windbreaks on 5% (7.45 Mha) of the cropland (25 Tg yr^{-1}). Despite many limitations including uncertainty of land areas under agroforestry, lack of standardized estimation protocols, and lack of accountability on various C stocks (source-sink services, detritus C, insect/pest damages, *etc.*), we believe these new accrual rates and the land areas under each practice are much more realistic as new information became available over the last decade.The total C sequestration by forests (776) and agroforests (219) is 995 Tg yr^{-1} and represents approximately 15% of the US CO_2 emissions. This review highlights the importance of sustainable management of forests and integration of agroforestry on agricultural lands to mitigate climate challenges further while meeting society's need for food and a healthy environment.

Keywords: Alley cropping, riparian buffers, silvopasture, windbreaks

INTRODUCTION

Climate change has been recognized as a major global threat affecting food security and sustainability. More greenhouse gases (GHG) are released into the atmosphere, and more landslides, floods, fires, and rising sea levels are being reported at an alarming rate. The world needs low-cost GHG reduction mechanisms that also guarantee various other benefits including food security, environmental stability, social acceptance, and better economies.

Management of agricultural and forestry systems to sequester C has been accepted as one of the cost-effective climate change mitigation strategies[1]. Establishing and maintaining healthy perennial vegetation that includes trees to enhance C sequestration (see [Box 1][2] for General Carbon Terminology used in this paper) is less expensive compared to most other techniques. Additionally, these practices come with other ecosystem services (ES) like enhanced water quality, soil health, land productivity, and social values with minimal environmental and health risks. Perennial vegetation allocates a higher percentage of C to belowground and often extends the growing season[1], which is more efficient than annual vegetation, and therefore further enhances the C sequestration potential of agriculture[3-6].

In this review article, we will focus primarily on two land use systems involving perennials, forests and agroforests, with great potential for C sequestration in the US. Trees and groups of trees identified as forests have the potential to positively impact climate change by reducing atmospheric carbon dioxide (CO_2) and providing long-term C storage. Trees and/or shrubs are combined with agronomic crops (annual or perennial) and/or livestock in agroforestry for various production, environmental, and economic benefits. Agroforestry offers greater potential to increase C sequestration of predominantly agriculture-dominated landscapes than monocrop agriculture[1,7-10]. Both these land use systems store C in above- and belowground biomass, soil, and living and dead organisms. Carbon can stay in these components for extended periods when managed sustainably.

FOREST C SEQUESTRATION

Forest area and extent

Worldwide, forests cover 31% of the global land area (4.06 billion ha)[11]. These forests offer many goods and services, including C sequestration. However, despite the United Nations targeting a 3% increase in forest cover by 2030, the forested area continues to decline. Although the rate of deforestation has slowed since 1990, the rate of forest expansion has also declined. During the past decade (2010-2020), the worldwide annual net loss of forest area was 4.7 million ha (Mha)[11].

The US has a total forested area of 310 Mha; globally, only three other countries have more areas under forest: the Russian Federation (815 Mha), Brazil (497 Mha), and Canada (347 Mha)[11]. These four countries

Box 1. General carbon terminology[2]

Carbon allocation	The total accumulation of carbon for a specified period
Carbon capture	See Carbon Sequestration
Carbon emissions	The amount of GHG or GHGs released into the atmosphere is often expressed by Carbon Equivalent. However, carbon dioxide is also considered a GHG
Carbon equivalent	The amount of carbon dioxide that, over time, produces a similar temperature change as a specified GHG or a mixture of GHGs. The conversion of specific GHGs to a measure of carbon
Carbon flux	The transfer, or change in amount, of carbon within a specific stock or pool. The balance of exchanges between what is sequestered and what is emitted from stock or pool of carbon
Carbon offset	The difference between Carbon Storage and Carbon Emissions. Often a term utilized in valuing and trading within economic sectors of society
Carbon pools	A particular location where carbon is stored or allocated for a period
Carbon sequestration	The process of storing carbon in a pool. Most often involving the process of fixation of atmospheric carbon by biological or physical processes
Carbon sink	A carbon pool where gains of carbon are greater than losses
Carbon stock	The amount of carbon in a specific pool
Carbon storage	See carbon stock

account for 54 percent of the world's forests[11]. Worldwide, forests have a dramatic influence on the environment. Obviously, the extent of this influence will decrease with the loss of forested land. From 1990 to 2020, the global area under forest decreased by 178 Mha, leading to a decrease in the total C stock in forests from 668 to 662 Gt (gigatons = 10*9 kg)[11]. Maintaining the global forests from further decline is critically important for maintaining a stable and healthy environment.

The US and UN (International) terminologies

It is important in this context to clarify and compare the terminologies used by the US agencies to report forest land area and C estimates. The US has been reporting on national resources since 1909 when the US Forest Service published *Timber Supply of the United States*. However, the Intergovernmental Panel on Climate Change (IPCC) began recommending reporting of GHG changes by land-use types[12]. In the most recent US accounting of forest C, the six land classifications are used to identify where C is located (forest land, cropland, grassland, settlements, wetlands, and other lands). Additionally, under the forest land categories, land use and land-use change have been used to clarify C capture and emissions between accounting periods. For instance, if land that was previously forested has now been converted to cropland, then its C impact will likely move from sequestration and sink to emitting C back to the atmosphere. This is accounted for by the IPCC guidelines and definition of land use, land-use change, and forestry (LULUCF)[13]. In general, C reporting on forest lands in the US is separated by LULUCF into the following land classifications: forest land remaining forest land, forest land converted to non-forest land, non-forest land converted to forest land, woodlands remaining woodlands, and urban trees in settlements[14,15].

In addition to GHG reporting, the worldwide perspective and correlation of forest resources require clarification. The use and definition of terms dealing with carbon and climate change that have generally been adopted and used around the world by the Food and Agriculture Organization of the United Nations (FAO) do not always align with those used by the US Forest Service (USFS) and other US agencies to describe our forest resources. The US forests have been described by data collected over time through fixed plots used in the Forest Inventory and Analysis program (FIA); this data predates worldwide tracking of climate change. Additionally, there is some disparity with other US agencies that collect environmental data. The US Environmental Protection Agency (EPA) utilizes data from the National Inventory Report (NIR), which, in addition to capturing data on GHG emissions and removals, reports data on land categorizations that include forestry. However, the NIR data often come from sources that use different definitions and time scales. Box 2[16] is a discussion on "forest" definitions from the *Forest Resources of the United States, 2017: a technical document supporting the Forest Service 2020 RPA Assessment*[17].

Within the US, too, the lack of standard historical definitions and reporting changes over time pose some challenges when reporting at the international level.

Forest trends

Forests and forested land areas are constantly changing because of natural processes and human activities. It is estimated that the forested area in the US area was reduced from 412 Mha (46% of the total land area) in 1630 to approximately 305 Mha (34% of the total) by 1910[16]. Change has often been ascribed to European settlement[16]. As such, the most noticeable reductions in forested land area occurred in the North and South regions of the US [Table 1]. By 2017, the forest land areas of the US had reduced to approximately 60% and 70% of the areas in 1630 in the North and South regions, respectively. By comparison, the Rocky Mountain and Pacific Coast regions retained more than 90% of their original forest cover estimates [Table 1].

While this change in land cover in some regions of the US is dramatic, since the early 1900s, the total forest area of the US has been relatively stable at around 300 Mha [Table 1]. This stability is reflected across most regions of the country. Still, a stable forested land area should not be misconstrued as never changing; these forested lands are constantly adapting and changing over time. For instance, Fei *et al.*[18] noted that during the 28-year period from 1980 to 2008, most areas of the eastern US experienced some decline in the abundance of oak (Quercus spp.). So, although the composition may change over time, the general definition of forest lands, i.e., land area under trees, is maintained.

Carbon sequestration

As a signatory to the United Nations Framework Convention on Climate Change (UNFCCC), estimates of C emitted from sources and stored in sinks have been recorded since 1994. Reporting requirements have identified that forests have a role to play in both emitting and storing carbon. Since lands and associated practices can both store and emit C, measurements are best captured as C flux. The C flux is the net of emissions and sequestration of GHG expressed in Tg CO_2 (teragram) equivalent (1Tg = 1MMT, which is commonly used in the US literature). A negative flux reflects sequestration of CO_2 equivalent in and by the forest ecosystem. Examples resulting in emissions of either form of C, or other GHG described as C equivalents, include when forested lands burn, when forest lands are converted to non-forest land, or when wood products are harvested. Examples that result in C storage include non-forest land converted to forest, harvested wood products, and forest land that remains forest land. Of note, while harvested wood products remove C from forest lands, a portion of that C is captured in long-term product and contributes to the C flux as sequestration[17]. In 2019 estimates of emissions and removals of C equivalent reflected 775.7 Tg captured by the US forest lands, including urban trees[15] [Table 1]. This is slightly lower than 2018 estimates, but somewhat higher than 2012 estimates. In general, the net flux from land use and land change has decreased since 1990.

In addition to the movement of C in and out of the forest, there is also accounting for how much C the forest system stores, referred to as C stock. In the forest ecosystem, C can accumulate in five storage pools[12] : aboveground biomass, belowground biomass, deadwood, litter, and soil C. The FAO generally apportions C stock as 44% in living biomass, 4% and 6% in dead wood and litter, respectively, and 45% in soil organic matter (SOM)[11]. Totaled annually, these storage pools highlight an accumulation and increase in C stock associated with forests. In 1990 total forest C stock on forest land remaining forest land in the US was

Box 2. How to define a forest[16]

Defining a forest
Defining forested land in a manner that is ecologically meaningful, nationally and internationally consistent, and measurable in the field is not always easy or straightforward. This report defines forest land in a manner consistent with the Food and Agriculture Organization of the United Nations (FAO) internationally agreed-upon definition: ***Forest land** — Land at least 120 feet (37 meters) wide and at least 1 acre (0.4 hectares) in size with at least 10 percent cover (or equivalent stocking) by live trees including land that formerly had such tree cover and that will be naturally or artificially regenerated. Trees are woody plants having a more or less erect perennial stem(s) capable of achieving at least 3 inches (7.6 cm) in diameter at breast height, or 5 inches (12.7 cm) diameter at root collar and a height of 16.4 feet (5 meters) at maturity in situ.* *In contrast, the domestic definition of forest land used by the Forest Inventory and Analysis (FIA) program of the Forest Service does not require trees to meet the in situ height requirement. Plots, where land is classified as "forest land" by FIA but are not productive enough to meet the FAO definition, have been placed into a category termed "woodland" for this report, which is defined as:* ***Woodland (FAO)** — Land at least 120 feet (37 meters) wide and at least 1 acre (0.4 hectares) in size with sparse trees capable of achieving 16.4 feet (5 meters) in height with a tree canopy cover of 5 to 10 percent combined with shrubs at least 6 feet (2 meters) in height to achieve an overall cover of greater than 10 percent woody vegetation. Trees are woody plants having a more or less erect perennial stem(s) capable of achieving at least 3 inches (7.6 cm) in diameter at breast height, or 16.4 feet (5 meters) at maturity in situ.* *Thus, forest and woodland categories in this report sum to match the FIA domestic forest land value. In addition to forest land and woodland, both programs (FIA and FAO) recognize timberland as a sub-classification of forest land, defined as:* *Forest land that is producing or capable of producing 20 cubic feet per acre or more per year of wood at the culmination of mean annual increment. Timberland excludes reserved forest lands.* *Prior to the 1990s, the United States collected data primarily on timberland. Therefore, in the interest of maintaining continuity with historical data, long-term trends in this report are often given for timberland instead of forest land. Definitions for other terms may be found in Glossary of Terms.*

estimated as 50,913 Tg[15]. If harvested wood and discarded wood product are added to this, then the total C stock would be 52,808 Tg C [Table 1]. From 1990 to 2019, total C stock increased, though the rate of increase seems to have slowed between 2017 to 2019 [Table 1]. From 1990 to 1995, total C stock rose by 2%, and from 2000 to 2005, it rose by 1.7%. The difference between 2017 and 2019 reflects only a 0.7% rise in total C stock. This may be partly explained by reductions in total forest area [Table 1]. For those same periods, the greatest reduction in total forest area was nearly 9 Mha. Nonetheless, it should be noted that, while total forest land is reduced for each of the aforementioned periods, total C stock continued to increase across each period.

CARBON LOSSES FROM THE US FORESTS

Forest management activities have major implications on C accounting, especially on soil organic C stock in forested environments, as highlighted in the review by Mayer *et al.*[19]. Disturbance and harvest activities are important factors that determine if forested lands are C sinks or sources, as well as levels of C stock. Williams *et al.*[20] identified that harvest was the most extensive disturbance in terms of area and carbon impacts, followed by fire, wind, bark beetles, and drought. They identified that living biomass in US forests collectively loses (flux) about 200 t (Mg) C y^{-1} from disturbance and harvesting. However, the total impact on C stock is estimated to be a positive increase of 190 Mg C y^{-1}[20]. The outcome of disturbance and management are not always clear and distinct from the responsive and resilient nature of the forest ecosystem.

Table 1. Total forest land (million ha) by year and region, and total forest CO_2 flux and carbon stocks for select time periods, United States

REGION	1630[1]	1907[1]	1953[1]	1963[1]	1977[1]	1987[1]	1990	1995	1997[1]	2000[2]	2005[2]	2012[1]	2017[1]	2018[2]	2019[2]
Total forest land (million hectares)	413.8	299.8	300.1	304.6	300.4	296.4	305.0	304.8	300.2	304.5	303.6	310.1	309.8	301.2	300.9
North	**120.4**	**54.4**	**65.1**	**67.1**	**66.4**	**66.9**	**-**	**-**	**68.9**	**-**	**-**	**71.1**	**71.1**	**-**	**-**
South	**143.3**	**100.1**	**97.2**	**99.1**	**95.3**	**94.7**	**-**	**-**	**93.5**	**-**	**-**	**99.0**	**99.4**	**-**	**-**
Rocky mountain region	**56.7**	**53.9**	**49.6**	**50.1**	**51.5**	**48.9**	**-**	**-**	**51.1**	**-**	**-**	**53.1**	**52.9**	**-**	**-**
Pacific coast	**93.5**	**91.4**	**88.3**	**88.3**	**87.2**	**86.1**	**-**	**-**	**86.7**	**-**	**-**	**86.8**	**86.4**	**-**	**-**
Total net flux -- Emissions and removals from land use, land-use change, and forestry in Tg CO_2 equivalent - negative reflects storage/sink[3]							(859.1)	(837.0)		(798.3)	(742.0)		(740.9)	(782.4)	(775.7)
Portion of net flux in living biomass[3]							(560.1)	(543.0)		(523.3)	(493.3)		(492.2)	(493.3)	(483.3)
Portion of net flux in soil (mineral and organic)[3]							0.9	0.2		(0.9)	(1.7)		16.4	(3.3)	(1.2)
Total carbon stock - above ground biomass, soil c-pools, harvested wood product (Tg C)[3]							52808.0	53870.0		54899.0	55842.0		58252.0	58443.0	58632.0
Portion of carbon stock in living biomass[3]							14129.0	14883.0		15613.0	16307.0		18097.0	18231.0	18363.0
Portion of carbon stock in soil (mineral and organic)[3]							31079.0	31078.0		31078.0	31081.0		31079.0	31079.0	31080.0

[1]Oswalt and Smith, 2019; [2]Oswalt and Smith, 2014; [3]Domke *et al.*, 2021.

Fire, insect, and disease

The impact of fire management practices on C stock can be either negative or positive. For instance, fire suppression seeks to remove fire from the landscape but then results in increased understory fuel loads and tree density - a positive increase in forest C stock[21]. However, as Mayer *et al.*[19] note, fire suppression has been shown to delay but not prevent wildfires. Currently, about 12% of the conterminous US forest land is at high or very high risk of wildfire due to suppression or other management that has created high fuel levels[16]. There is cause for concern since fire directly releases C back into the atmosphere, but the complete picture of this is not always clear because fire also results in C being stored long-term on the site through tree mortality and charred wood, and it usually results in a pulse of new regeneration.

In general, wildfires have been documented to have a strong negative effect on forest soil C. Wildfires also impact aboveground biomass C stocks. From 1997 to 2008, the fire burned between 1.4 and 2.7 Mha annually in the conterminous US, with approximately half of that area being forestlands[20,22]. Using FIA data, remote sensing, and published biomass data, Gonzalez *et al.*[23] estimated aboveground C stock for the state of California, USA, and identified that wildland fires in 6% of the state's forest area accounted for two-thirds of the loss in C stock from 2001 through 2010. Fuel loads initially increase the C stock of forested lands, but where fire suppression fails, the result is often greater fire intensity. Once fire releases C, recovery can take many years. Depending on several factors (e.g., fuel moisture, fire intensity, fire duration, tree species, etc.), forest C stock may take from 10 to 128 years to recover[19] from a major fire incident.

In summarizing research on post-fire C stock recovery, Williams *et al.*[20] noted that forest regrowth, though it may take 20-100 years to balance emissions, will often do so within historically typical fire regime frequencies. Therefore, the C outcome from fires is often neither good nor bad; instead, it is time-reference dependent. However, practices that reduce fire intensity and/or duration may preserve more of the on-site C stock identified with soils and living aboveground biomass[19].

Across the US, less than 1% of standing inventory mortality is due to insects and diseases[16], although the evidence of this is very location-dependent. The authors noted that while mortality in the South has declined since 2006, pine mortality in the west has dramatically increased as a result of the mountain pine beetle. Williams *et al.*[20] estimate that insects and diseases could transfer as much as 21 million tons (Tg) of live aboveground biomass to litter and woody debris pools on an annual basis. Hicke *et al.*[24] identified that from 1997-2010, beetles killed trees that contained 2-24 Tg C y^{-1}, but that more trees had been killed since 2000 than in earlier periods. Williams *et al.*[20] noted that other species of trees adjacent to those impacted by insects or diseases will then often see compensatory increases in growth. These considerations represent the challenges associated with predicting single-year impacts on C, either stock or flux.

Harvest

Annual harvesting occurs across the US on approximately 4.5 Mha, of which 61% is by selective harvesting and 39% is clearcutting[16]. Clearcutting is the most common forest harvesting practice worldwide and generally has negative consequences for soil C, while partial or selective harvesting methods tend to reduce soil C loss[19]. Noormets *et al.*[25], in a review of forest management on productivity and C sequestration, point out that forests may assist with climate change mitigation and provide wood products, but maximizing merchantable productivity may also have a high cost on soil C. However, the accounting for soil C in the US forests shows that C stock remained mostly constant from 1990-2019, and soil continued, by and large, to contribute towards C removal from the atmosphere by flux [Table 1].

Timber harvests typically remove stem wood and place that wood into a product where C is stored for some amount of time; they do not result in an immediate release of all biomass C[15]. While the harvesting of wood products does not in itself create a C sink, if the C resides in the wood product for longer than the time it takes for forest regrowth to recover C removal or loss due to harvesting, then the net effect on atmospheric C will appear as sequestration[20]. Simply stated, wood residue, or discarded product, that is taken to a disposal site tends to retain and keep a portion of C locked up for many years. From a high of 123.8 Tg CO_2 equivalent in 1990 to a low of 69.1 in 2010, the total flux accounted for by harvest wood has varied[15]. Most recently, the 2019 flux total associated with harvested wood is 108.5 Tg CO_2 equivalent and includes disposed wood products. As a percent of the total next flux of the C pool associated with forest lands remaining forest lands, harvest wood totals reflect 15.7%, 10.1%, and 15.7%, respectively, for the years 1990, 2010, and 2019[15]. For the same years, harvest wood totals are only 3.6, 4.3, and 4.6% of the total C pool stock. The harvested wood product does not generally seem to detract from either the net flux of atmospheric C or the longer-term storage pool of C stock.

PROJECTIONS IN US FORESTS CARBON STORAGE POTENTIAL

Projections in forest C are just that, projections. They are models based on assumptions, such as timber consumption, wood for energy use, timber harvest, and timber prices, all over an extended number of years. Most model projections focus on 2050 or 2060 to define outcomes based on their assumptions. However, these models and assumptions often provide necessary input for climate change policy development.

When projecting the US forest C sequestration for the period from 2010 - 2060, Nepal *et al.*[26] indicated that projected capacities could be "notably altered" by the wood energy sector of the economy. Growth in variables such as the use of woody biomass for energy and new forest plantations in the southern US had a large impact on model outcomes and could either predict US forests becoming an emissions source or significantly increase additional storage[26]. Their model highlights the sensitivity of forest C sequestration outcomes associated with forest management, forest product utilization, and policy decisions.

Recognizing the value of the impact of policy on the C sequestered by US forests, Haight *et al.*[27] evaluated several scenarios and suggested two which offer the greatest opportunity to increase C sequestration through 2050. The authors compared three policy scenarios against a baseline scenario: "(1) a land-use policy to reduce deforestation from development; (2) an afforestation policy targeting rural landowners in the eastern US and a reforestation policy targeting understocked federal forest lands in the western US; and (3) a policy reducing stand-replacing fire events by 10-percent." Their model emphasized two results: the change in C sequestration and an estimated dollar value of that change. Results identified that the greatest C gains and monetary values would be generated by policy scenario 2, followed by 1, and then 3[27]. Their scenario analysis highlights that policy impacts C sequestration and monetary gains, and that afforestation and reforestation offer the greatest positive outcomes by 2050[27].

Mitigating climate change is strongly tied to landowners' willingness to participate, policy-driven decisions, and market values. In the carbon economy, the relationship of these variables to decision processes is challenging to correlate. Nonetheless, the interrelated nature of carbon price, policy, and landowners' willingness to participate will continue to direct how nations participate and act towards improving the health of our climate.

AGROFOREST C SEQUESTRATION

Agroforestry has been recognized as having the greatest potential for C sequestration of all the land uses analyzed in the LULUCF report of the IPCC (2000). However, our understanding of C sequestration in specific agroforestry practices from around the world is rudimentary at best. The best available estimates for the US come from a synthesis published by Udawatta and Jose[28]. We will provide an update on their numbers in this review. Of the recognized agroforestry practices in the US, C sequestration by alley cropping, riparian buffers, silvopasture, and windbreaks are presented in this chapter, while forest farming and urban food forests are not included as data is limited.

Alley cropping

Alley cropping consists of a widely spaced single or multi-species tree, grass, and/or shrub rows with agronomic crops or pasture grass grown in the alleys[29]. Tree and crop row configuration, differences in C input into the soil, decomposition rate, previous management, and associated soil microfauna determine spatial, temporal, above-, and belowground heterogeneity in C stock and sequestration rates all influence C flux within the agroforestry system[30,31].

The available literature on alley cropping C sequestration is provided in Table 2. The data range from 0.01 to 96.5Mg ha^{-1} for aboveground and 2.5 to 77.1 Mg ha^{-1} for soil C sequestration, depending on species, age and spacing of the tree component. For example, in a 5-year-old alley cropping in northeast Missouri, pin oak (*Quercus palustris* Muenchh), bur oak (*Q. macrocarpa* Michx.), and swamp white oak (*Q. bicolor* Willd.) on a corn (*Zea mays* L.)-soybean (*Glycine max* L.) rotation, trees sequestered 0.05 Mg C ha^{-1} in five years (76 trees ha^{-1}[32]). In Georgia, *Albizia julibrissin* (mimosa) alley cropped with grain sorghum (*Sorghum bicolor*) during summer and wheat (*Triticale aestivum*) grown over winter, sequestered 50 times more C than the 5-year-old Missouri study. The tree density was 2,400 ha^{-1} at 0.5-m spacing within rows and 4-m spacing between rows.

Greater C sequestration potential of litter material of alley cropping has been reported by several Canadian and US studies. In Guelph, Canada, C inputs through litterfall on a poplar-spruce alley cropping with wheat (*Triticum aestivum* L.)-soybean-maize rotation were 0.6 and 0.95 Mg C ha^{-1} in the 11th and 12th years[33,34]. In a 6-year-old hybrid poplar (111 trees ha^{-1}) study, Thevathasan and Gordon[35] reported that 1.07 Mg C ha^{-1} was contributed by litterfall. At the same site, Peichl *et al.*[36] reported 1.3 and 5.5 Mg C ha^{-1} for hybrid poplar leaves and branches when trees were 13-year-old. At age 13, trees and litter+fine roots added 14 Mg C ha^{-1} and 25 Mg C ha^{-1}[37].

A limited number of studies show greater root mass and C in alley cropping than in monocropping. In Indiana, the root biomass of red oak (*Q. rubra* L.) was 2.1 and 1.8 times greater than the maize root biomass at the tree base and 1.1 m from the base[38]. A 47-year-old pecan (*Carya illinoinensis*)-cotton (*Gossypium hirsutum*) alley cropping system in Florida had 1.75 Mg C ha^{-1} (398 mg C kg^{-1} soil) compared to 0.38 Mg C ha^{-1} (88 mg C kg^{-1}) in the 3-year-old system[7]. In a recent study in Missouri, Salceda *et al.*[39] showed greater soil C accumulation in the alley cropping than in row crop areas. Soil C increased from 1.89% in 2000 to 2.19% in 2020 for tree rows compared to 1.85% to 1.93% in crop areas for 0-10 cm soils. These values translate to 4,244 kg C ha^{-1} (212 kg C ha^{-1} y^{-1}) increase in tree rows and 1,991 kg ha^{-1} (99 kg C ha^{-1} y^{-1}) in the crop areas. Tree rows, grass waterways, and crop alleys showed a 16.5, 9, and 5% C increase in the top 10-cm soils over 20 years, respectively [Figure 1].

Riparian buffers

Riparian areas are complex terrestrial assemblages of plants and other organisms adjacent to an aquatic environment. These include the transition zones between upland and aquatic habitats such as wetlands, streams, rivers, lakes, and bays[40,41].

There are a number of studies available related to C sequestration by riparian buffers. Available data summarized in Table 2 vary widely. The aboveground C of a mature riparian forest ranged from 50 to 150 Mg ha^{-1}[42]. Another riparian system in Washington, US, also showed a similar biomass accumulation pattern with an increase in C from 9 to 271 Mg ha^{-1} as the system matured (age ~ 250-year). Almost 90% of the stem density and biomass accumulation occurred during the first 20-40 years[42].

Comparing C sequestration among riparian, crop, pasture, and also with the system age, Tufekcioglu *et al.*[43] reported four and eight times greater aboveground C for poplar areas (~20 Mg ha^{-1}) of the riparian buffer compared to 5 and 2.5 Mg C ha^{-1} for switchgrass (*Panicum virgatum* L.) and cool-season grass areas in Iowa [Figure 2]. Adjacent corn and soybean areas had 3 and 1.3 Mg ha^{-1} aboveground C.

In addition to the C sequestered in roots, riparian soils store C in soil organic matter, which contains about 50% C. Mature riparian stands hold greater amounts of SOM compared to monocropped agroecosystems or

Table 2. Biomass (above and below), soil, and microbial carbon stocks of various agroforestry practices at different locations in the United States (since agroforestry-specific data is limited, certain Canadian locations are also included)

Agroforestry practice	Location	Age (years)	Species/treatment	C ($Mg\ ha^{-1}$)† Above or Belowground‡	Soil	Microbial	Source
Alleycropping	Missouri	5	Pin oak Bur oak Swamp white oak	0.03, 0.01, 0.015			Udawatta *et al.* 2005[32]
	Missouri	22	Pin oak Bur oak Swamp white oak	8.65 2.79 4.90			Udawatta *et al.* Unpublished
	Georgia	1	Mimosa tree mulch with grain sorghum and winter wheat	2.5			Rhoades *et al.* 1998[78]
	Guelph, Ontario, Canada	15	poplar intercrop spruce intercrop barley sole crop	96.5 75.3 68.5			Peichl *et al.* 2006[36]
	St. Remi, Quebec, Canada.	8	tree-based conventional systems		77.1 (0-30)* 43.5 (0-30)		Bambrick *et al.* 2010[31]
	Guelph, Ontario, Canada	21	Poplar Norway spruce conventional systems		57 (0-30) 51 (0-30) 51 (0-30)		Bambrick *et al.* 2010[31]
	Guelph, Ontario, Canada	15	poplar intercrop spruce intercrop barley sole crop		3.0 (0-20) 2.5 (0-20) 2.4% (0-20)		Peichl *et al.* 2006[36]
	Florida	3 3 47 47	Pecan orchard Pecan-cotton Pecan orchard Pecan-cotton			1.2% 1.9% 4.3% 3.4%	Lee and Jose 2003[7]
	Florida	3 47	pecan system pecan system			0.38 0.78	Lee and Jose 2003[7]
Riparian buffers (Aboveground)	Washington	~ 250	N/A	9 to 271			Balian and Naiman 2005[42]
	North Carolina	>50 25-50 5-25 0-5		156 60 61 2.2	46 (0-10) 35 (0-10) 12.5 (0-10) 39 (0-10)		Rheinhardt *et al.* 2012a[79]
	Iowa		Poplar Switchgrass Cool season grasses	20 5 2.5			Tufekcioglu *et al.* 2003[43]
	South Carolina	2 8 12 60	N/A	< 7.5 17.5, 17.5, 106			Giese *et al.* 2003[45]
	Northeast Ontario, Canada	95		29.3-269.1			Hazlett *et al.* 2005[80]
	Iowa	6	Poplar-switchgrass-coolseason grass	35			Tufekcioglu *et al.* 1999[81]
	California	30	Riparian woodland Cottonwood-willow Upland riparian Mixed riparian Willow scrub	99.6 202 156 95 86			Matzek *et al.* 2018[82]
		100	Riparian woodland Cottonwood-willow Upland riparian Mixed riparian Willow scrub	237 215 180 95 87			
	Quebec	6	Poplar 3570 Poplar 3230 Poplar 915508	11 kg $tree^{-1}$ 13 13			Fortier *et al.* 2010[83]

			Poplar915311 Poplar3729	15 18		
	Ontario		Sugar maple and Ash Cedar and Pine	247 100		Vijayakumar *et al.* 2020[84]
Riparian buffers (Belowground)	Iowa	6	Poplar Switchgrass Coolseason	6 9 7		Tufekcioglu *et al.* 1999[81]
	South Carolina	2 8 12 60	N/A	2.5 3.7 5 5.5	4.2% (0-15) 4.7% (0-15) 4.0% (0-15) 11.4% (0-15)	Giese *et al.* 2003[45]
	South Carolina	>50 25-50 5-25 0-5			39 (0-10) 34 (0-10) 28 (0-10) 42 (0-10)	Rheinhardt *et al.* 2012a[79]
	New York			0.25 to 14.4 mean 6.6		Kiley and Schneider 2005[85]
	Iowa	6	Poplar Switchgrass Coolseason grass Crop (soybean)		2.4 (0-35)† 1.8 (0-35)† 1.8 (0-35)† 0.4 (0-35)†	Marquez *et al.* 1999[44]
	Iowa.	7-17	Riparian buffer Warmseason grass Coolseason grass		50.2 (0-15) 47.2 (0-15) 55.3 (0-15)	Kim et al. 2010[46]
	Iowa	16-26	Riparian buffer Warmseason grass Coolseason grass Corn-soybean		70.8 (0-15) 56.2 (0-15) 57.8 (0-15) 57.1 (0-15)	Kim *et al.* 2010[46]
	Iowa	7	Riparian		60 (0-15)	Kim *et al.* 2010[46]
	Central Ohio		Restored riparian		33 (0-15)	Marton *et al.* 2014[86]
			Natural riparian		25.7 (0-15)	
	New Jersey		Riparian		100 (0-30)	Bedison *et al.* 2013[87]
	Quebec, Canada		Woodlot Herbaceous buffer Poplar		120 (0-60) 92 (0-60) 85 (0-60)	Fortier *et al.* 2013[88]
	Ontario, Canada	60 6	Riparian Deciduous riparian Conifer riparian		193 (0-60) 5.94% (0-60) 2.87% (0-60)	Vijayakumar *et al.* 2020[84]
Silvopasture	Oregon		Pastures agroforestry plantation	0 12.24 6.95		Sharrow and Ismail 2004[89]
	Oregon		*Understory C* Pastures agroforestry plantation	1.0, 1.17 2.23		Sharrow and Ismail 2004[89]
	Oregon		Pastures agroforestry plantation		102.5 (0-45) 95.9 (0-45) 91.94 (0-45)	Sharrow and Ismail 2004[89]
	Florida		Pasture Center of alley between tree row		1033 (0-125) 1376 (0-125) 1318 (0-125)	Haile *et al.* 2010[51]
	Arkansas	17	Pecan silvopasture	0.75 Mg $ha^{-1}yr^{-1}$		Dold *et al.* 2019[50]
			Oak silvopasture	0.2 Mg $ha^{-1}yr^{-1}$		
	Southcentral	0-25	Pine silvopasture	1.5 Mg $ha^{-1}yr^{-1}$		Lee and Dodson 1996[90]
		25-50		0.31 Mg $ha^{-1}yr^{-1}$		

	Florida	22	Pine bahiagrass		69 (0-30)	Adewopo et al. 2015[53]
Windbreak	Canada		By species	See Table 3		Kort and Turnock 1999[58]
	Canada	17-90	Conifer single row	24-41 Mg km^{-1}		
			Poplar single row	105 Mg km^{-1}		
			Shrub single row	11 Mg km^{-1}		
	Nebraska		Conifer single row	9.14 t km^{-1}		Brandle *et al.* 1992[63]
			Hardwood single row	5.41 t km^{-1}		
			Shrub single row	0.68 t km^{-1}		
	Nebraska	35	Windbreak crop field		39.94 (0-15) 36.23 (0-15)	Sauer *et al.* 2007[91]
	Nebraska	50	Conifers	1.07-3.84		Possu *et al.* 2016[59]
			Hardwood	0.99-13.6		
	Northeast	50	Single row	0.3		
	Appalachian	50	Three-row	5.8		
	Rockymount	50	Three-row	0.8 km^{-1} yr^{-1}		
	Delta region	50	10-row	12.7 km^{-1} yr^{-1}		
	Northeast	50	One-row	0.5		
	Corn belt	50	One-row	7.7		
	S. Dakota	19			1.9 (0-100)	Chendev *et al.* 2015[92]
	Great plains				183 (0-125)	Khaleel *et al.* 2020[61]
	Saskatchewan	18 20 22 27 31 32	Caranga Green ash Manitoba maple White spruce Hybrid poplar Scot pine		79.5 (0-50) 149.3 (0-50) 106.5 (0-50) 111.6 (0-50) 143.4 (0-50) 124.5 (0-50)	Dhillon and Van Rees 2017[62]

†In Mg ha-1 yr-1; ‡assumed 50% C in the biomass to estimate C when C concentration was not provided. Unless specified values represent aboveground C; *Soil depth in cm in parenthesis.

younger riparian buffers[44,45]. A poplar-grass riparian system established in 1990 in Central Iowa showed 1.2 and 0.9 Mg C ha^{-1} yr^{-1} C accrual rates for poplar and switchgrass zones[44]. In South Carolina, Giese *et al.*[45] observed 2.6 times greater soil organic carbon (SOC) in a 60-year-old buffer compared to 2-, 8-, and 12-year-old riparian buffers. Kim *et al.*[46] studied riparian buffer soils to a 15 cm depth in Iowa and showed a SOC increase of 50 to 71 Mg ha^{-1} in seven years, representing a 29% increase.

Riparian buffers can serve as a sink as well as a source for some GHGs. Riparian areas showed significantly higher CO_2 emissions than in adjoining croplands (6.8 ± 0.6 *vs.* 3.6 ± 0.5 Mg CO_2-C ha^{-1} y^{-1})[47]. Flooding is strongly correlated with CH_4 emission (up to + 44.5 mg CH_4-C m^{-2} d^{-1}), especially under warm soil conditions (> 22°C). However, the effect was less pronounced in early spring (< 1.06 mg CH_4-C m^{-2} d^{-1}) due to low soil temperature. Annual CH_4 emission for croplands and the flood-affected riparian forest was +0.04 ± 0.17 and +0.92 ± 1.6 kg CH_4-C ha^{-1}, respectively. The non-flooded riparian zone served as a net sink (-1.08 ± 0.22 kg CH_4-C ha^{-1} y^{-1}) due to better irrigation, while a depression (< 8% of the total area) in the riparian zone accounted for 78% of the annual CH_4 emission. In another study, Jacinthe and Vidon[48] showed that riparian buffers act as a net sink for CH_4. Riparian buffers exhibited -0.80 to -0.34 mg CH_4-C m^{-2} d^{-1} uptake rates.

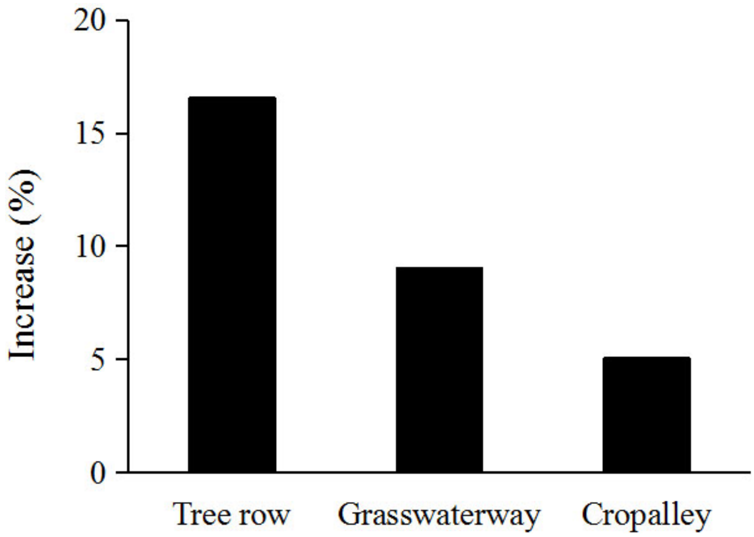

Figure 1. Percent increase in soil organic carbon in the 0-10 cm soil depth between 2000 and 2020 for tree rows, grass waterways, and crop alleys at the paired watersheds at the Greenley Research Center, Knox County, Missouri[39].

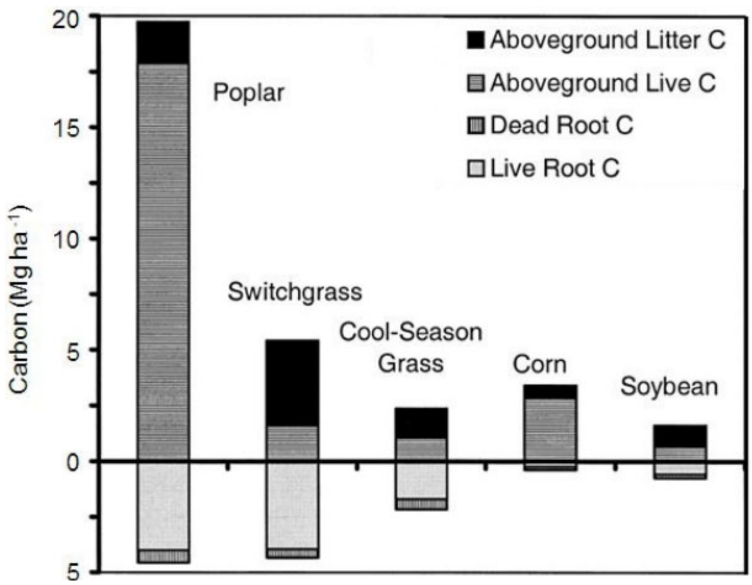

Figure 2. Litter and root carbon distributions in a riparian system with trees, grass, and crops in Iowa, USA[43].

Silvopasture

Silvopasture is the most common form of agroforestry in North America[49]. In silvopasture, trees, forage crops, and livestock are intentionally integrated into a structural and functional system for optimization of benefits from planned biophysical interactions.

The C distribution in above- and belowground, and soil varies spatially depending primarily on the tree configuration, age and species of trees used [Table 2]. For example, in Arkansas, 17-year-old tree woody biomass and C values were 7.1 and 3.4 Mg ha^{-1} for pecan (*Carya illinoinensis*) and 26.6 and 12.7 Mg ha^{-1} for oak (*Quercus rubra*) for silviculture management[50]. Pecan and oak trees sequestered C at the rate of 0.75 and 0.20 Mg C ha^{-1} yr^{-1}, respectively. In Florida, open pasture, center of the pasture alley, and in-between trees in tree row had 1033, 1376, and 1318 Mg SOC ha^{-1} to a 1.25-m depth, respectively[51]. Silvopastures contained more C in deeper soil layers and stored more stable C than pastures mainly because of the dead and live deep roots of trees[51,52]. Silvopastures almost doubled mineral-associated C fractions than rangeland (5.7 *vs.* 10 g C kg^{-1}[53]). Another study in Central Missouri showed significantly greater C and roots in soils under a cottonwood (*P. deltoides* Bortr. ex Marsh.)-pasture than corn-soybean[54,55].

Land conversion to silvopasture increases C accumulation compared to pure pastures or rangeland[53,56]. For example, silvopasture (69.2 Mg C ha^{-1}) contained 1.12 and 1.69 times more soil C in the 0-30 cm soil than sown pasture (62 Mg C ha^{-1}) and native rangeland (40.9 Mg C ha^{-1}) 22 years after the conversion from native rangeland to silvopasture or sown pasture[53].

Windbreak

Windbreaks contain one or more rows of trees or shrubs to reduce wind speed, evaporation, wind erosion and blowing snow, protect homes, structures, livestock, and crops, provide habitat for wildlife, improve landscape, and mitigate odor[57]. Windbreak vegetation sequesters C directly by storing C in their biomass and indirectly by reducing energy use and enhancing greater C assimilation in the companion vegetation and soil[58,9]. Windbreaks are not new to the US. The Prairie State Project in the 1930s planted 223 million trees from Texas to North Dakota (okhistory.org/learn/depression3) to protect and improve soils and land productivity.

Using windbreak field data from Nebraska and Montana, 15 allometric models, and destructive sampling, Possu *et al.*[59] estimated C storage potential of 1.07 ± 0.21 to 3.84 ± 0.04 Mg C ha^{-1} yr^{-1} for conifer species (mean 2.45 ± 0.42 Mg C ha^{-1} yr^{-1}) and 0.99 ± 0.16 to 13.6 ± 7.72 Mg C ha^{-1} yr^{-1} for broadleaved deciduous species (mean 4.39 ± 1.74 Mg C ha^{-1} yr^{-1}) during a 50-year period [Table 2]. Carbon storage potentials of different windbreak designs ranged from 0.3 Mg C km^{-1} yr^{-1} for a single-row small-conifer windbreak in the Northeast region to 5.8 Mg C km^{-1} yr^{-1} for a three-row tall-deciduous windbreak in the Appalachia region[60].

In a recent study, Khaleel *et al.*[61] reported increases of 16% C beneath tree windbreak plantings in the US Great Plains. Windbreak soils of 0-125 cm depth stored 183.2 Mg C ha^{-1}, which was 29.4 Mg C ha^{-1} larger than the adjacent fields[61]. Soil C to 50-cm depth ranged from 79 Mg ha^{-1} in 18-yr-old Caranga to 149.3 Mg ha^{-1} in 20-yr-old Green Ash Shelterbelt species in Saskatchewan, Canada, [Table 2][62].

The North Central region needs 94 Mha of windbreaks to reduce crop damages[63]. The country also needs windbreaks to protect homes, farm structures, and roads, livestock fencing, visual screening, aesthetics, and odor mitigation. For example, 450,000 concentrated animal operations need windbreaks for odor mitigation and visual perception.

CHALLENGES TO ESTIMATING C SEQUESTRATION IN AGROFORESTRY

During the last 40 years numerous studies have evaluated C sequestration of agroforestry in the USA[2,10,32,38,52,58,63]. Many research studies have been established to fulfill specific objectives of the study or to meet the requirements of the funding/granting agency. There is tremendous heterogeneity among studies in terms of the study design, establishment, management, soil sampling depth, replication, data collection, measured parameters, reported results, durations of studies, and record keeping. For example, Udawatta *et al.*[64] evaluated soil C to a 1-m depth for samples collected in 2000 by landscape positions, while Salceda *et al.*[39] followed a grid sampling design and collected only the surface 20-cm soils in 2020 to compare temporal changes in soil C at the same watersheds. Missing parameters like soil bulk density, soil classification, and other required parameters restrict conversions from percentage C to mass; thus, results are incomparable. These differences restrict estimations for above- and belowground biomass, and soil C by practice and to a reasonable soil depth.

Similarly, information on trees such as species, tree density, age, height, dbh (diameter at breast height, measured at 1.37 meters above groundline), and spacing as well as crop, pasture, and livestock even within a major soil group and/or a climatic region cannot be used to estimate C for practices due to large differences among studies or lack of information. Herbicide, fertilizer, tillage, and other management practices with rates, dates, forms and daily weather data should be available for comparisons and conversion of values to facilitate comparisons. Additionally, land management history before the conversion to agroforestry should be provided or available in a data repository to explain changes during the study. Differences in the method of soil sampling, preparation of samples for analysis, and methods of analysis form another major group of factors that can influence the results substantially. Thus, available datasets are mostly incomparable, incomplete, and heterogeneous, which seriously hinder rigorous comparisons among studies and their conclusions.

In agroforestry, tree volume and mass are usually estimated using forestry tree growth models. However, open-grown trees in agroforestry typically have different architecture compared to narrowly spaced forest-grown trees. Trees in agroforestry hold lower branches and leaves for extended periods, thus contributing to more biomass and C in the aboveground. Trees in agroforestry usually have larger dbh, larger crowns, and greater tapering than forest plantations. According to Zhou *et al.*[65], trunk biomass was underestimated from 6.3% to 16.6% with forest growth models applied to open-grown agroforestry trees.

Uncertainty on the land area under agroforestry is a major challenge in estimating potential C sequestration. Dependable estimates in the absence of accurate statistics of areas under each agroforestry practice are required to estimate the C sequestration potential of a country or region. Zomer *et al.*[66] estimated global agricultural lands with tree covers by remote sensing at 1 km^2 resolution. A similar procedure can be implemented, and land area can be estimated with greater accuracy using high-resolution images, remote sensing, drone imaging, and newer image analyzing technologies.

Agroforestry is a young discipline, and as such, databases for various important parameters that influence C estimations are still missing in the literature. Most studies did not account for C contributions of woody debris, standing dead trees, and O-horizon in their C estimations. Although leaves and litter inputs contribute to C accumulation, those are not available across regions, practices, and age groups. Greenhouse gas emission and source-sink services of agroforestry are available only for a handful of studies. Such information on temporal and spatial changes is required to determine C dynamics and net gains to develop climate-friendly and profitable agroforestry design for all climatic, soil, and geographic regions.

Starting from February 2022, National Agricultural Statistics Service (NASS) of USDA (United States Department of Agriculture) (https://www.nass.usda.gov/Surveys/Guide_to_NASS_Surveys/Agroforestry/index.php accessed January 29, 2022) will be conducting the first-ever National Agroforestry Survey, which will collect information on alley cropping, riparian forest buffers, silvopasture, windbreaks, as well as forest farming and multi-story crops. Information on the type of agroforestry, the number of practices, the way the species are used for the selected practice, and some additional information will be collected. This will provide the initial basic information to evaluate existing agroforestry practices, such as their extent, age, and tree-crop-animal combinations at each location. This is a significant step forward in inventorying agroforestry at the national level in the US.

We have excluded data on forest farming and urban food forests, and thus, our estimates are incomplete and underestimate the C sequestration potential of agroforestry. Hopefully, the new census by USDA may help resolve this and some other challenges to some extent. Finally, we wish to acknowledge that the focus of this chapter is on the agroforestry practices in the USA. Various other forms and types of agroforestry practices that are managed in multiple ways for a large array of products and services exist around the world.

PROJECTIONS OF C SEQUESTRATION POTENTIAL OF AGROFORESTRY

According to the USDA[67], the cropland area in the US was 149 Mha in 2017. The total alley cropping practices on cultivated cropland, non-cultivated cropland, and pastureland were 22,600, 16,000, and 14,300 ha in 2007, 2012, and 2017[67]. Only 0.009% (13,840 ha) of the cultivated and non-cultivated cropland had alley cropping in 2017. There is a decline in the alley cropping land area and most of the decrease was in cultivated cropland. During the same period, contour buffer strips were practiced on 435,775 and 17,321 ha of cultivated and non-cultivated cropland[67]. The total alley cropping and contour buffer practices in 2017 were 211,938 ha (0.14% of the cropland). Approximately 80 Mha of cropland[68,8] and 40 Mha of highly erodible nonfederal land[29] are available and suitable for alley cropping in the US. This represents 15% of the total US landmass or 54% of the cropland.

Practical adoption of alley cropping and contour buffer strips is very limited, and we assumed only 0.5% of the cropland may establish these practices for our calculations. Our mean aboveground and soil C accruals were 26 and 41 Mg ha^{-1} [Table 4]. The total accumulation was 67 Mg ha^{-1} and we considered a 10-year period for the accumulation; thus, annual rate of sequestration was 6.7 Mg ha^{-1} yr^{-1}. Our review does not provide sufficient information to estimate reliable values for above- and belowground, and soil C accurately. Therefore, we used the mean value 0.5 Mg ha^{-1} yr^{-1} (0.13, 0.479, 0.5, and 0.922) for the C sequestration rate on 0.5% of the cropland (745,000 ha) to calculate annual C sequestration on US croplands. This value (0.37 Tg yr^{-1}) is several magnitudes lower than values estimated by others. This difference is mainly caused by the small land area used in our estimation, as this land area is more realistic and practically achievable. Although the adoption rate of alley cropping is very low, if 5% (7.5 Mha) of the cropland adopts alley cropping, the C sequestration potential of alley cropping on US cropland could be as large as 3.7 Tg C annually.

Our mean above- and belowground, and soil C accruals for riparian buffers were 99, 5.9 and 81 Mg ha^{-1} [Table 4]. The total accumulation in those three components was 185.9 Mg ha^{-1}. The age of studies in this analysis ranged from 2 to 150 years and the mean was 33 years. The C sequestration of 5.6 Mg ha^{-1} yr^{-1} was estimated by dividing the total accumulation (185.9 Mg ha^{-1}) by the mean age (33 yr). The total measured riparian forest buffers in the US were 640,723 ha[67]. The total river and stream length in the US is approximately 5.65 million km[69]. Buffer width of 15 m on both sides of streams was sufficient to protect water and provide habitat for wildlife in Nebraska[70]. For the protection of levees, buffers should be wider than 91-m[71]. The width and the composition vary with landowner objectives and expected functions[72]. If a 15-m wide riparian buffer is established along both sides of 5% of the total river length, it will occupy 847,500 ha. This area is only 24% larger than the 2017 riparian area estimated by the USDA[67]. Using a conservative estimate of 5.6 Mg C ha^{-1} yr^{-1} accrual rate, the potential C sequestration by riparian buffers along rivers in the US could be approximately 4.746 Tg C yr^{-1}. In this calculation, smaller water bodies were excluded, and thus, the C sequestration estimation is lower than the true potential.

In the US, 248 Mha of permanent pastures/range, 51 Mha of grazed forests, and 40 Mha of unmanaged forests may offer the potential for conversion to silvopasture and accrue more C. In some regions, grazing

Table 4. Estimated C sequestration potential in aboveground and belowground parts and soil for major agroforestry practices in temperate North America

		C Stock†			C
Practice		Minimum	Maximum	Mean	Sequestration Rate‡
		----------- Mg C ha^{-1} -------------			Mg C ha^{-1} yr^{-1}
Alley cropping	Aboveground	0.01	96.5	26	
	Belowground				
	Soil	0.05	77	41	
					6.7
Riparian Buffers					
	Aboveground	2.2	247	99	
	Belowground	0.25	14.4	5.9	
	Soil	1.8	386	81	
					5.6
Silvopasture	Aboveground	1.17	12.2	4.9	
	Belowground				
	Soil	1.03	1.38	1.21	
					6.1
		------------- Mg C km^{-1} ------------			Mg C ha-1 yr^{-1}
Windbreaks	Aboveground	0.68	105		
	Belowground				
	Soil		23.1		6.4
				kg tree^{-1}	Mg ha-1 yr^{-1}
	Hybrid poplar			367	0.73
	White spruce			186	0.37

†This analysis used published data for the United States and Canada as reported in Table 2. If not given, we assumed 50% C in the above and belowground biomass to estimate C stocks; ‡harvest age of 50-year was assumed for riparian buffers. Harvest age of 20-year and tree density of 40 tree ha-1 were assumed to estimate annual C accrual rates for windbreaks on cropland.

also occurs either on marginal lands or as a secondary activity on high-yielding timberlands, which also can be converted to silvopasture. Based on our review, silvopastoral systems appear to sequester 6.1 Mg C ha^{-1} yr^{-1}. Using a sequestration potential of 6.1 Mg C ha^{-1} yr^{-1} on 10% marginal pastureland (25 Mha), grazed forests (5 Mha), and unmanaged forests (4 Mha), the total C sequestration potential for silvopastoral lands in the US could be as high as 207.1 Tg C yr^{-1}.

The total land area under windbreaks, shelterbelts, and living snow fences in the USA has increased significantly during the last 15 years [Figure 3]. We calculated the mean annual C sequestration for windbreaks of mixed-species by taking the values of conifers and hardwood from Possu *et al.*[60] as 3.42 Mg C ha^{-1} for a 50-year cycle. Carbon sequestration of 2.37 Mha of windbreaks is 8.1 Tg C yr^{-1}. The total C sequestration potential of windbreaks could be 25 Tg C yr^{-1} if 5% (7.45 Mha) of the cropland is converted to windbreak and sequesters C at a 3.42 Mg C ha^{-1} rate. Carbon sequestration of current windbreak lands (2.37 Mha) is 8.1 Tg C yr^{-1} at the same rate.

The total C sequestration of current alley cropping (211,938 ha), riparian buffers (640,732 ha), silvopasture (34 Mha), and windbreak (2.37 Mha) areas can be estimated as 219 Tg C yr^{-1} (0.37+3.58+207+8.1) based on their estimated respective accrual rates [Table 4]. Using those same accrual rates, total agroforestry C sequestration could be increased to 240 Tg C yr^{-1} if (1) 5% of the US cropland could be converted to alley cropping (3.7 Tg yr^{-1}); (2) 15-m wide riparian buffers on both sides of 5% of the total stream length

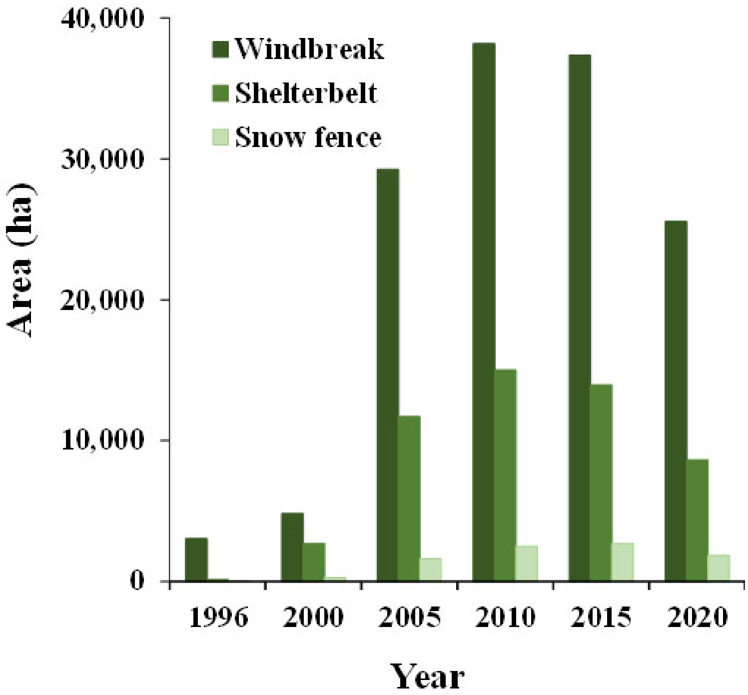

Figure 3. Annual establishment of windbreaks, shelterbelts, and living snow fences from 1996 to 2020 in the USA (http://usda.library.cornell.edu/concern/publications/j3860694x?locale=en&page=3#release-items Accessed December 5, 2021).

established (4.75 Tg yr^{-1}); (3) 34 Mha silvopasture could be established on grazing lands (207 Tg yr^{-1}); and (4) windbreaks could be expanded on to 5% (7.45 Mha) of the cropland (25 Tg yr^{-1}). The total C sequestration potential estimated by this review is 40% (219 Tg yr^{-1}) of the previous estimate (548 Tg yr^{-1}) by Udawatta and Jose[28]. This difference is mainly due to the decrease in the land area we considered under alley cropping, riparian buffers, and windbreak practices for this revised review. Both the accrual rates and the land areas under each practice are much more realistic as new information became available over the last decade.

CARBON ECONOMY

In the US and North America, there are two primary carbon markets. They are the Western Climate Initiative (which includes California and Quebec) and the Regional Greenhouse Gas Initiative (11 states primarily in the Northeastern US). These operate with carbon pricing mechanisms that fall into three main categories: Cap-and-Trade (i.e., Emissions Trading Systems (ETS)), carbon taxation, or hybrid mechanisms that combine elements of both[73]. They emphasize the trade of CO_2 emissions. Most ETS work from the concept that an emitter of C and GHG will have a level set for desirable emissions. According to the UNFCCC, a baseline can either be defined for an emitter as a reasonable representation of emissions that would occur in the absence of CDM (Clean Development Mechanisms) emissions limitation and reduction commitments, or it can be defined for a potential carbon sink as the "sum of the changes in C stocks in the C pools within the project boundary that would occur in the absence" of an approved CDM project[74]. Whether an emitter or a sequestration project, its baseline is defined as where your C would be in the absence of any change by an approved UNFCCC project. This baseline standard, usually referenced by date in time, became the basis around which carbon accounting is conducted for an entity or land area.

If a carbon emitter is legally required to reduce emissions, one way to do that is to purchase carbon credits to offset their emissions levels. Similarly, if an emitter wishes to increase emissions above a legally established baseline, then they can subsidize equivalent emission reductions by purchasing stored C and thus offset their emissions. The idea of forest carbon offsets was first proposed at COP13 in Bali through Reducing Emissions from Deforestation and Forest Degradation (REDD+, with a plus added to represent afforestation efforts)[75]. Offsets reflect the difference between C storage and actual emissions. The offset, or measurable reduction in GHG, has monetary value and may be tradeable in carbon markets[76]. However, for forest landowners to enroll in carbon markets, it must be shown that projects (activities or management) on their lands will increase net GHG (carbon) removals above levels that would have otherwise occurred[74]. This concept is defined by the UNFCCC as additionality.

When the baseline is coupled with concepts and application of offset and additionality, much debate ensues related to the actual impact that marketing of carbon offsets has on mitigating climate change. While certain states within the US have compliance with carbon markets (e.g., California), most of the US forests make carbon offsets available through voluntary carbon markets[76]. Do these carbon markets result in additional emissions reductions? That question depends on how forest carbon is measured and accounted for. Baseline and additionality both create the opportunity for differences of opinion related to the impact on atmospheric C. For instance, Cames *et al.*[77] identified that 85% of the EU projects under the UN's CDM failed to reduce emissions. As Gifford[75] points out, there is power and authority in how C is defined and managed.

CONCLUSION

The forests and agroforests of the US, indeed worldwide, are highly appreciated for their role in the reduction of atmospheric CO_2. For example, over the past few decades, the world's forests have absorbed almost the same amount of CO_2 emissions as the ocean. IPCC has recognized agroforestry as having the highest C sequestration potential of any land management system. In general, the process of afforestation will increase the land area sequestering C to durable C sinks. However, there are also challenges to afforestation if land area is taken away from food production. Therefore, new approaches to increasing the land area in trees should explore opportunities that merge with other societal needs such as food production and environmental conservation, for which agroforestry is a solution. Whether in net atmospheric flux, or longer-term C stock, trees help increase C density. Based on our analysis of the existing data, the annual C sequestration by forests and agroforests is 776 Tg yr^{-1} and 219 Tg yr^{-1}, respectively. The total C sequestration (995 Tg yr^{-1}) by these two land use systems represents approximately 15% of the US GHG emissions. Therefore, the greater the land area that is occupied by trees, the greater the impact on atmospheric C. Although C sequestration data by various agroforestry practices and the land areas occupied by these practices are not readily available, along with forests, agroforests, and special agroforestry applications help mitigate climate challenges while also meeting society's need for food and a healthy environment.

Table 3. Above and belowground biomass and carbon for shelterbelt trees commonly used in Saskatchewan, Canada[58]

		Above-ground	Below-ground	Total C
			------------- kg tree^{-1} -------------	
Deciduous	Green ash	161.8	64.7	110
	Manitoba maple	178.6	71.4	120
	Hybrid poplar	544.3	217.7	367
	Siberian elm	201.9	80.8	140
Conifers	White spruce	286.9	86.1	186
	Scot pine	164.1	49.2	107
	Colorado spruce	202.2	60.7	131
Shrubs	Choke cherry	402.6	201.3	302
	Villosa lilac	334.6	167.3	251
	Buffaloa berry	312.0	156	234
	Caranga	516.0	258	387
	Sea buckthorn	213.0	106	160

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Authors' contributions

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All authors declared that there are no conflicts of interest.

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Not applicable.

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