Carbon Sequestration Potential of Agroforestry Practices in Temperate North America

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Abstract Agroforestry, an ecologically and environmentally sustainable land use, offers great promise to sequester carbon (C). The objectives of this chapter are to (1) provide a review of C sequestration opportunities available under various agroforestry practices in temperate North America, and (2) estimate C sequestration potential by agroforestry in the US. Since accurate land area under agroforestry was not available, the potential C sequestration was estimated based on several assumptions about the area under different agroforestry practices in the US: 1.69 million ha under riparian buffer, 17.9 million ha (10% of total cropland) under alley cropping, and 78 million ha under silvopasture (23.7 million ha or 10% of pasture land and 54 million ha of grazed forests). Based on these, we estimated C sequestration potential for riparian buffers, alley cropping, and silvopasture in the US as 4.7, 60.9, and 474 Tg C $year^{-1}$, respectively. Establishment of windbreaks to protect cropland and farmstead could sequester another 8.79 Tg C $year^{-1}$. Thus, the potential for C sequestration under agroforestry systems in the US is estimated as 548.4 Tg $year^{-1}$. The C sequestered by agroforestry could help offset current US emission rate of 1,600 Tg C $year^{-1}$ from burning fossil fuel (coal, oil, and gas) by 34%. These preliminary estimates indicate the important role of agroforestry as a promising CO_2 mitigation strategy in the US, and possibly in other parts of North America. The analysis also reveals the need for long-term C sequestration research in all

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regions and for all agroforestry practices, establishment of standardized protocols for C quantification and monitoring, inventory of agroforestry practices, development of models to understand long-term C sequestration, and development of agroforestry design criteria for optimum C sequestration for all regions.

Keywords Alley cropping • Riparian buffers • Silvopasture • Windbreaks

Introduction

Rising levels of atmospheric carbon dioxide (CO_2) and associated global warming have moved to the center stage of climate change discussion in the past two decades. While many dispute the global warming hypothesis, projected doubling of atmospheric CO_2 by the latter half of the Twenty-first century raises concerns for everyone. Significant reductions in the atmospheric CO_2 concentrations can only be achieved with substantial additional costs and major changes in living standards. Therefore, adoption of CO_2 reduction strategies are widely debated, not well received, and not agreed upon by all nations. The world needs carbon (C) sequestration techniques that provide social, environmental, and economic benefits while reducing atmospheric CO_2 concentration.

Management of agricultural systems to sequester C has been accepted as a partial solution to climate change (Morgan et al. 2010). Establishing and maintaining perennial vegetation to enhance C sequestration is less costly compared to most other techniques, and these practices have minimal environmental and health risks. Perennial vegetation is more efficient than annual vegetation as it allocates a higher percentage of C to below-ground and often extends the growing season (Morgan et al. 2010), therefore enhancing C sequestration potential of agricultural systems even further (Lal et al. 1999; Watson et al. 2000; Oelbermann et al. 2006a; Jose 2009).

Agroforestry, as a system that combines trees and/or shrubs (perennial) with agronomic crops (annual or perennial), offers great promise to sequester C both above- and below-ground. Agroforestry practices have been approved as a strategy for soil C sequestration under afforestation and reforestation programs and also under the Clean Development Mechanisms of the Kyoto Protocol (Watson et al. 2000; IPCC 2007; Smith et al. 2007). Adoption of agroforestry practices has greater potential to increase C sequestration of predominantly agriculture dominated landscapes than monocrop agriculture (Lee and Jose 2003; Nair and Nair 2003; Nair et al. 2009; Schoeneberger 2009; Morgan et al. 2010). Within agroforestry systems (AFS), C can be stored in above- and below-ground biomass, soil, and living and dead organisms. The quantity and quality of residue supplied by trees/shrubs/grass in agroforestry systems enhance soil C concentration (Oelbermann et al. 2006b). In addition, C stored by trees could stay in soils or as wood products for extended periods of time. If agroforestry systems are managed sustainably, C can be retained in these systems for centuries (Dixon 1995). The amount of C stored on a site is a balance between long-term fluxes. However, the net C gain depends on the C content of the previous system that the agroforestry practice replaces (Schroeder 1994).

The enhanced C sequestration concept is based on efficient use of resources by the structurally and functionally more diverse and complex plant communities in agroforestry systems compared to sole crop or grass systems (Sanchez 2000; Sharrow and Ismail 2004; Thevathasan and Gordon 2004; Steinbeiss et al. 2008; Marquez et al. 1999). Agroforestry practices accumulate more C than forests and pastures because they have both forest and grassland sequestration and storage patterns active (Schroeder 1994; Kort and Turnock 1999; Sharrow and Ismail 2004). For example, an alley cropping system with pine trees and pasture grass could efficiently utilize light energy at different canopy levels compared to a monocropping system. Species in agroforestry systems often have different physiological needs for particular resources in certain amounts, at certain times, and possess different structural or functional means to obtain them (Jose et al. 2004). The utilization of the environment by species includes three main components: space, resources, and time. Any species utilizing the same exact combination of these resources as another will be in direct competition which could lead to a reduction in C sequestration. However, if one species differs in utilization of even one of the components, for example light saturation of C3 vs. C4 plants, C sequestration will be enhanced.

Although agroforestry has come of age during the past three decades and scientific data has expanded, our understanding of C storage and dynamics in AFS is still minimal (Nair et al. 2010). Similarly, a complete picture of C distributions in AFS in the North American Continent is lacking in the literature, thus restricting development of suitable mitigation strategies to enhance C sequestration associated with establishment of agroforestry practices on the agricultural landscape (Udawatta and Godsey 2010). Reliable estimates of soil C sequestration are essential for development of management plans related to climate change (Watson et al. 2000). This is especially important in AFS due to their complex nature, differences among practices, climatic conditions, and site factors. Well designed long-term research is needed to fill the knowledge gap so that appropriate agroforestry systems could be developed to maximize C sequestration benefits (Reed 2007). The objectives of this chapter are to (1) provide a review of C sequestration opportunities available under various agroforestry practices in temperate North America, and (2) synthesize available data and estimate C sequestration potential by agroforestry in the US.

Data Collection and Analysis

A literature search was conducted to identify peer-reviewed papers and government documents pertaining to agroforestry related C sequestration in five major temperate agroforestry practices namely; riparian buffers, alley cropping, silvopasture, windbreaks, and forest farming (Table 1). Scientific conclusions on C storage and sequestration as influenced by management practice and other factors were included in the analyses. Studies on C sequestration were categorized by practice (Table 1). Forest farming was not included since sufficient information was not available for an in depth review. When C concentrations were not provided, biomass was assumed

Table 1 Five main agroforestry practices in temperate North America

Practice	Predominant region and distribution	Function
Riparian and upland buffers	All regions	Ameliorate non point source pollution Protect watersheds and stream banks
Silvopasture	West and Southeast; all regions	Economic diversification Improve animal health Create wildlife habitat Fire protection Timber management
Alley cropping	Midwest; all regions	Increase bio diversity Increase income
Windbreaks	Great plains; all regions	Protect crop, animal, and structures Enhance crop and animal production Control erosion Distribute snowfall
Forest farming	All regions	Diversify income

Source: Gold and Garrett (2009). Reproduced with permission

to contain 50% C. Although literature from both the US and Canada were reviewed, the combined data set was used to estimate overall C sequestration potential of agroforestry practices in the US only.

Riparian Buffers

Riparian areas have many definitions which vary with the intended function and geographic region, but are generally defined as a complex terrestrial assemblage of plants and other organisms adjacent to an aquatic environment (Table 1). These include the transition zone between upland and aquatic habitats such as wetlands, streams, rivers, lakes, and bays. They are linear in shape and characterized by laterally flowing water that rises and falls at least once within a growing season (Lowrance et al. 1985; Welsh 1991).

Riparian systems store C in above- and below-ground biomass of the vegetation and in soils. Biomass accrual varies by region, plant composition, soil, climate, age, and management. The diverse species mixture of riparian buffers helps enhance C sequestration potential spatially and temporally compared to monocropping systems. The different functional groups such as trees and grasses in these systems colonize and capture both the above- and below-ground resources more effectively than the row crop agriculture.

In general, C sequestration potential and storage are greater in the above-ground portion of riparian buffer systems compared to row crops or upland forests. In riparian systems, tree density and basal area are often greater than or equal to those of upland

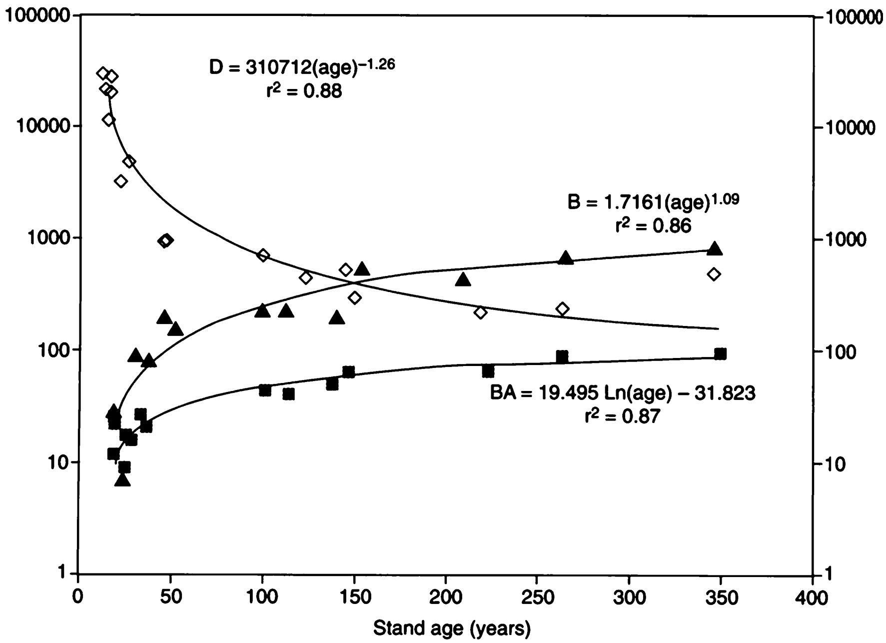

Fig. 1 Changes in stem density (D, stems ha^{-1}), biomass (B, Mg ha^{-1}), and basal area (BA, m^2 ha^{-1}) of a riparian forest buffer with age in Washington, USA. Y axis in logarithmic scale for stems ha^{-1}, Mg ha^{-1}, and m^2 ha^{-1} (Source: Naiman et al. 2005. Reproduced with permission)

forests due to prevailing favorable growth conditions. Above-ground C of a mature riparian forest ranged from 50 to 150 Mg ha^{-1} (Naiman et al. 2005). As riparian systems mature, the above- and below-ground biomass of the understory and overstory vegetation increase, giving an overall increase in the system level C stock. According to Naiman et al. (2005), stem biomass accrual of a riparian forest buffer can be determined by stand age (stem biomass kg ha^{-1} = 1.7161* age$^{1.09}$; r^2 = 0.86). Stem biomass accrual and thereby C stock increased at a diminishing rate for stands < 150 year and reached a plateau after 250 year (Fig. 1). Biomass accumulation pattern of another riparian system in Washington, USA, showed similar patterns 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 (Table 2; Balian and Naiman 2005).

Similar observations were made by Boggs and Weaver (1994) and Harner and Stanfoord (2003) in Montana, USA. In their study, willow (*Salix* spp.) and cottonwood (*Populus deltoides* Bortr. ex Marsh.) riparian buffers developed into a mature system (~ 60 year) where above-ground C increased from 0.5 to 97 Mg ha^{-1} while stem density decreased from > 10,000 to < 1,300 stems ha^{-1}. Tufekcioglu et al. (2003) observed four and eight times greater above-ground C for poplar areas (~20 Mg ha^{-1}) of the riparian buffer in Iowa compared to 5 and 2.5 Mg C ha^{-1} for switchgrass

Table 2 Biomass (above and below), soil, and microbial carbon stocks of various agroforestry practices at different locations in temperate North America

Agroforestry practice	Location	Age (years)	Species/treatment	C (Mg ha^{-1}) Above-ground[a]	Below-ground	Soil	Microbial	Source
Riparian buffers	Washington, USA	~250	N/A	9–271				Balian and Naiman (2005)
	Iowa, USA		Poplar	20				Tufekcioglu et al. (2003)
			Switchgrass	5				
			Cool season grasses	2.5				
	South Carolina, USA	2	N/A	<7.5	2.5			Giese et al. (2003)
		8		17.5	3.7			
		12		17.5	5			
		60		106	5.5			
	Northeast Ontario, Canada	95		29.3–269.1				Hazlett et al. (2005)
	Iowa, USA	6	Poplar+	35	6			Tufekcioglu et al. (1999)
			Switchgrass+		9			
			Cool season grass		7			
	New York, USA			0.25–14.4 mean 6.6				Kiley and Schneider (2005)
	Iowa, USA	6	Poplar			2.4		Marquez et al. (1999)
		6	Switchgrass			1.8		
		6	Cool season grass			1.8		
		1	Soybean			0.4		
	Iowa, USA	7–17	Riparian buffer			50.2		Kim et al. (2010)
			Warm season grass			47.2		
			Cool season grass			55.3		
	Iowa, USA	16–26	Riparian buffer			70.8		As above
			Warm season grass			56.2		
			Cool season grass			57.8		
			Corn-soybean			57.1		

Alley cropping	Missouri, USA	5	Pin oak	0.03			Udawatta et al. (2005)
			Bur oak	0.01			
			Swamp white oak	0.015			
	Georgia, USA	1	Mimosa tree mulch + grain sorghum + winter wheat	2.5			Rhoades et al. (1998)
	Guelph, Ontario, Canada	15	Poplar intercrop	96.5	3.0		Peichl et al. (2006)
			Spruce intercrop	75.3	2.5		
			Barley sole crop	68.5	2.4%		
	St. Remi, Quebec, Canada.	8	Tree-based conventional systems		77.1		Bambrick et al. (2010)
					43.5		
	Guelph, Ontario, Canada	21	Poplar		57		As above
			Norway spruce		51		
			conventional systems		51		
	Florida, USA	3	Pecan orchard			1.2%	Lee and Jose (2003)
		3	Pecan + cotton			1.9%; 0.38	
		47	Pecan orchard			4.3%	
		47	Pecan + cotton			3.4%; 0.78	
Silvopasture	Oregon, USA		Pastures	0	102.5		Sharrow and Ismail (2004)
			Agroforestry	12.24	95.9		
			Plantation	6.95	91.94		
	Oregon, USA		*Understory*				As above
			Pastures	1.0			
			Agroforestry	1.17			
			Plantation	2.23			
	Florida, USA		Pasture		1033		Haile et al. (2010)
			Center of alley		1376		
			Between tree row		1318		
Windbreak	Nebraska, USA	35	Windbreak		39.94		Sauer et al. (2007)
			Crop field		36.23		

[a]Assumed 50% C in the biomass to estimate C when C concentration was not provided

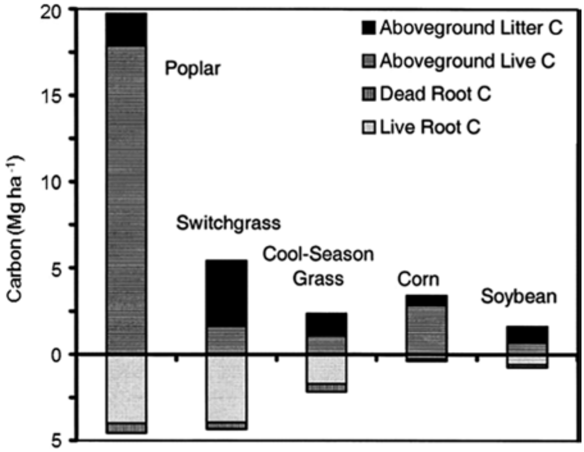

Fig. 2 Litter and root carbon distributions in a riparian system with trees, grass, and crops in Iowa, USA (Source: Tufekcioglu et al. 2003. Reproduced with permission)

(*Panicum virgatum* L.) and cool season grass areas, respectively (Fig. 2). Adjacent corn (*Zea mays* L.) and soybean [*Glycine max* L. (Merr.)] areas had 3.0 and 1.3 Mg ha^{-1} above-ground C, respectively. Giese et al. (2003) reported 106 Mg ha^{-1} C in a 60 year-old riparian buffer compared to <7.5, 17.5, and 17.5 Mg ha^{-1} in 2, 8, and 12 year-old buffers, respectively in South Carolina (Table 2). The total amount of C (including roots, herbs, and shrubs) stored in the mature riparian forest buffer in this study was four times that of the younger stands. Studying C storage in riparian (0–5 m from the water body) versus upslope forested area (60–75 m from the water body) in northeastern Ontario, Canada, Hazlett et al. (2005) observed 3% more C in the riparian zones (Table 2).

The aforementioned studies provide a realistic estimate of above-ground C stock of mature riparian buffer systems in temperate North America. If the system is maintained, these values may reflect the sequestration potential at maturity and would allow estimates of annual accrual rates. From these data, we estimate, for mature riparian buffers, an average above-ground C stock of 123 Mg C ha^{-1} for a 50 year cutting cycle. The estimated average above-ground C sequestration potential is 2.46 Mg C ha^{-1} year^{-1} (Table 3). In Canada, C accruals of 29–269 Mg ha^{-1} were reported for riparian buffers (Hazlett et al. 2005). Using published data (n = 4), Schroeder (1994) estimated 63 Mg C ha^{-1} above-ground storage for temperate zone riparian buffers with a 30 year cutting cycle. Our mean estimate of 123 Mg C ha^{-1} is twice the value estimated by Schroeder (1994). According to Hoover and Heath (2011), above-ground C stock for forest stands could range from 74 to 106 Mg ha^{-1} with a mean of 90 Mg ha^{-1}. Riparian areas are generally highly productive and therefore the value could be greater than for a typical forested site.

Roots of the riparian buffers also sequester significant quantities of C below-ground and this C is retained in the soil C pool as roots decay. Studying root densities in riparian-crop transects in Iowa, Tufekcioglu et al. (1999) found significantly greater root biomass in the riparian vegetation compared to the row-crop areas (Fig. 3). On average, poplar (*P. deltoides* x *nigra* DN-177), switchgrass, and cool season grass root C during the study were 3, 4.5, and 3.5 Mg C ha^{-1}, respectively (Table 2; Fig. 3).

Table 3 Estimated C sequestration potential in above-ground and below-ground vegetation parts and soil for major agroforestry practices in temperate North America

Practice		C Stock[a] (Mg C ha^{-1}) Minimum	Maximum	Mean	C sequestration rate[b] (Mg C ha^{-1} year^{-1})
Riparian buffers	Above-ground	7.5	269	123	2.6
	Below-ground	2.0	14.4	4.6	
	Soil	1.8	5.5	3.6	
Alley cropping	Above-ground	0.05	96.5	26.8	3.4
	Soil	0.05	25	6.9	
Silvopasture	Above-ground	1.17	12.2	4.9	6.1
	Soil	1.03	1.38	1.21	
Windbreaks[c]	Above-ground	0.68	105		
	Soil		23.1		6.4
	Hybrid poplar[d]			367	0.73
	White spruce[d]			186	

[a]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 below-ground biomass to estimate C stocks

[b]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

[c]C Stock in windbreaks s expressed as Mg C km^{-1}

[d]Mean C stock for hybrid poplar and white spruce are in kg tree^{-1}

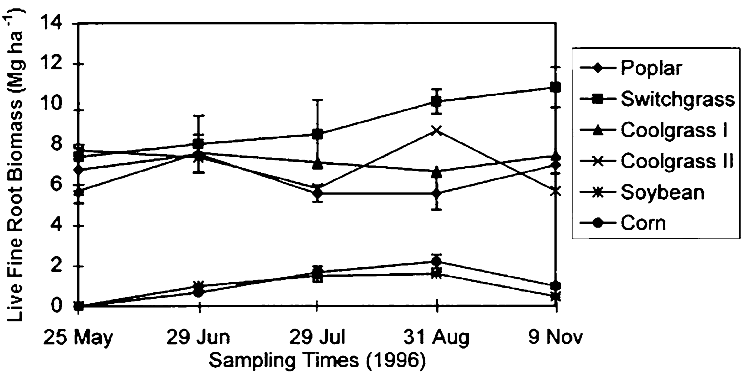

Fig. 3 Fine root biomass of trees, grasses, and crops in a riparian-row crop continuum in Central Iowa, USA (Source: Tufekcioglu et al. 1999. Reproduced with permission)

The riparian vegetation consisting of trees and grasses also had more fine (0–2 mm dia.) and medium (2–5 mm dia.) sized roots in the surface soil and throughout the 165 cm soil profile. Coarse and medium roots of poplar trees in the riparian zone extended beyond 165 cm depth while no crop roots were found below 125 cm. Four years after establishment, root biomass and thereby below-ground C were significantly greater in the riparian zone than the row crop areas (Marquez et al. 1999).

Another study, also from Iowa, demonstrated the potential to sequester greater quantities of C in soils under riparian buffers compared to row crops. Below-ground C in the tree and switchgrass areas of the riparian buffers was significantly greater

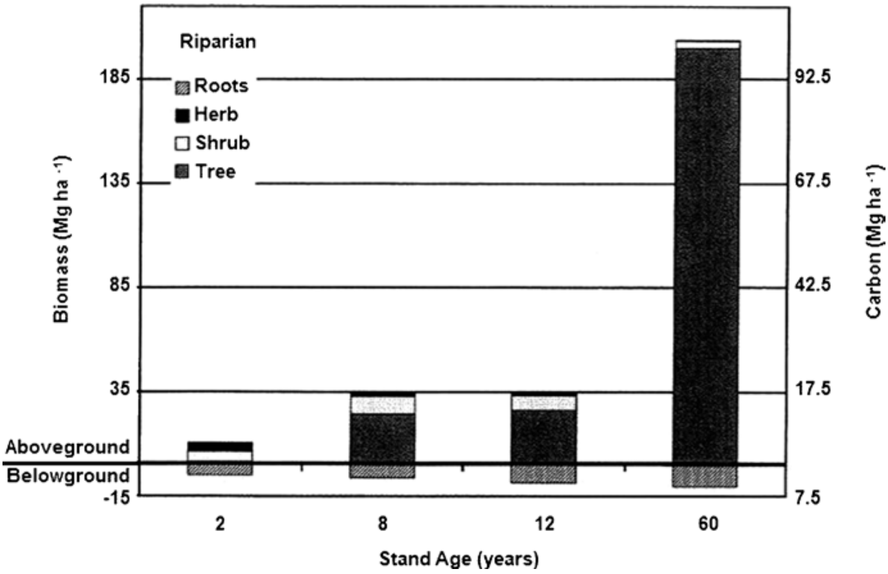

Fig. 4 Above- and below-ground biomass and carbon of 2-, 8-, 12-, and 60-year-old riparian stands in South Carolina, USA (Source: Giese et al. 2003. Reproduced with permission)

than accompanying grasses and adjacent corn-soybean crop areas (Tufekcioglu et al. 2003). Figure 2 shows >4.5 Mg C ha^{-1} for poplar and switchgrass areas of the riparian buffer compared to <2 Mg C ha^{-1} for cool season grass and <1 Mg C ha^{-1} for corn and soybean. Similar below-ground C accrual rates were reported by Giese et al. (2003) in South Carolina. The results showed 2.5, 3.7, 5.0, and 5.5 Mg C ha^{-1} below-ground in 2, 8, 12, and 60 year-old riparian buffers, respectively (Fig. 4). This study also showed that fine root biomass in the younger stands was 25–50% of that found in mature stands. In the Adirondack Park, New York, root C of riparian buffers was between 0.25 and 14.5 Mg ha^{-1} with a mean of 6.6 Mg ha^{-1} (Kiley and Schneider 2005). Other studies on root C of riparian buffers reported values ranging from 1 (Jones et al. 1996) to 3 Mg ha^{-1} (Tufekcioglu et al. 1999). Greater root mass of mature riparian stands indicate the importance of long-term management of riparian buffers for enhanced C sequestration.

The aforementioned results were used to estimate the below-ground C sequestration potential of riparian buffers (Table 3). Root biomass and C accumulation follows an asymptote as in above-ground biomass and C accumulation with an early increase in accumulation rates and a plateau as the system matures (Giese et al. 2003). We estimated a mean C sequestration of 0.09 Mg C ha^{-1} year^{-1} in below-ground tissues for riparian buffer systems for a 50 year harvest cycle.

In addition to the C sequestered in roots, riparian soils store C in soil organic matter (SOM). The SOM, which contains about 50% C, is greater in mature riparian stands compared to monocropped agroecosystems or younger riparian buffers (Giese et al. 2003). A riparian system consisting of poplar trees and grasses established in Central Iowa (Marquez et al. 1999) in 1990 showed significantly greater total soil C

Table 4 Soil carbon and soil organic matter percentage in 2-, 8-, 12-, and 60-year-old riparian buffer systems in South Carolina, USA

Stand age (year)	Soil carbon (%)	Soil organic matter (%)
2	4.2	12.3
8	4.7	12.9
12	4.0	10.7
60	11.4	30.3

Source: Giese et al. (2003). Reproduced with permission

concentrations in 1991 and 1996 than the nearby crop areas. In the poplar and switchgrass zones, soil organic carbon (SOC) accrued at the rate of 1.2 and 0.9 Mg C ha^{-1} $year^{-1}$. The results of this study introduce a promising and important observation: changes in soil C can occur in a relatively short period of time and therefore establishment of riparian buffers with the appropriate vegetation combination may help sequester soil C in a short period at a low cost. Giese et al. (2003) observed that soil C content was 2.6 times greater in the 60 year-old buffer in South Carolina compared to 2, 8, and 12 year-old riparian buffers (Table 4). Kim et al. (2010) studied riparian buffer soils to a 15 cm depth in Iowa and showed a SOC increase of 50–71 Mg ha^{-1} in 7 years, representing a 29% increase (Table 2). Warm season and cool season grass buffers contained 47 and 56 Mg C ha^{-1} in the sampled 15 cm soil depth.

The litter material in the riparian zones, either from the riparian vegetation or flooding, also contributes to soil C sequestration. The litter is approximately 47% of the above-ground biomass production (Piedade et al. 2001). The litter production rate is inversely related to the latitude of the riparian systems (Benfield 1997). In general, riparian buffers with infrequent flooding produce 5.5 Mg ha^{-1} litter material (Piedade et al. 2001); however, this varies by vegetation type (Tufekcioglu et al. 1999). Riparian buffers that are frequently or permanently flooded produce less litter than infrequently flooded buffers (Conner et al. 1981; Piedade et al. 2001).

One of the factors determining net soil C sequestration is soil respiration. In Iowa, annual soil respiration rates within a riparian buffer and adjacent crop field varied between 7.4 and 12.2 Mg C ha^{-1} $year^{-1}$ (Tufekcioglu et al. 2001) with the highest in the streamside cool season grass buffer and the lowest in the corn areas. Annual respiration rates were 11.5, 11.4, 10.3, and 7.5 Mg C ha^{-1} $year^{-1}$ for crop side cool season grass, poplar buffer, switchgrass, and soybean, respectively. Although the perennial vegetation in the buffer areas (trees or grasses), had significantly greater respiration rates compared to the annual crops, trees leaf out and grasses begin to grow before the crop is established and, thus increasing the overall C sequestration potential of the system (Marquez et al. 1999).

Management Implications

As the above literature reveals, riparian buffer systems have tremendous potential to sequester C in above- and below-ground plant parts and in the soil compared to monocropping systems in temperate North America. These systems sequester C in a relatively short period of time and the sequestration rates are high during the early stages of development. Root C deposited in the deeper horizons of the soil profile could remain for extended periods, and thereby contributing to long-term soil organic C storage. Proper management operations such as maintenance of suitable buffer width along water bodies, proper species selection, and removal of older trees following best management practices (BMPs) would further enhance C sequestration potential of riparian systems.

Management agencies at both national and state levels have determined appropriate buffer dimensions for protection of water bodies. For example, the Massachusetts Department of Conservation and Recreation specifies that riparian buffer width should increase by 12 m for every 10% increment of slope greater than 10%. In general, buffer widths between 15 and 100 m have been suggested by various local, state, and federal agencies for water quality improvements, stream bank stabilization, and to reduce sediment and nutrient losses (NRCS 2007). Buffers of 91 m width have been proposed for levee protection and to stop flooding and other damages (Dwyer et al. 1997). The composition and the width of the riparian buffer system also vary and much wider buffers are required for the removal of soluble nutrients compared to stabilizing stream banks (Schultz et al. 2009). Multi-species riparian buffers could consist of fast and slow growing trees adjacent to the water body and shrubs and grasses between the trees and upland areas. Properly designed riparian buffers not only sequester C, but improve water quality, wildlife habitat, aesthetic value, economic returns, and land value (Qiu and Prato 2001; Schultz et al. 2009).

The total river and stream length in the US is approximately 5.65 million km (3.533 million miles; USEPA 2010). Lakes and estuaries occupy 16.8 million ha and 22.7 million ha, respectively. The nationwide NRCS goal was to establish 3.2 million km (two million miles) of conservation buffers by 2002 (http://www.nrcs.usda.gov/feature/buffers/BufrsPub.html#InitiativeBuff_7Anchor, accessed 15 January 2011). Documented goals or committed riparian buffer lengths vary by state. For example, Chesapeake Bay agreement for riparian buffers fulfilled their 960 km riparian buffer target for 2010 (http://www.unl.edu/nac/insideagroforestry.htm, accessed 24 January 2011). If a 30 m wide riparian buffer is established along both sides of 5% of total river length it would occupy 1.69 million ha. Using a conservative estimate of 2.6 Mg C ha^{-1} year^{-1} accrual rate (Table 3), the potential C sequestration by riparian buffers along rivers in the US could be as high as 4.7 Tg C year^{-1}. This approximation ignored smaller and/or intermittent streams that were not part of the total river length as well as other water bodies that do not have a measurable perimeter for the estimation of buffer length. Some of these water bodies currently have riparian buffers established for various ecological and environmental reasons. Other water bodies that have disconnected buffers or no buffers offer a greater C sequestration potential through establishment of riparian buffers.

The 4.7 Tg year^{-1} C sequestration potential estimated by this analysis is significantly greater than the 1.5 Tg C year^{-1} estimated by Nair and Nair (2003). This difference is due to the area considered for the sequestration and the values used to estimate the sequestration potential. Nair and Nair (2003) used 30 m wide forested riparian buffer zone on one fourth of the 3.2 million km conservation buffers committed by the USDA in 2002 for their estimate.

Alley Cropping

Alley cropping has received increased attention in temperate North America in recent years. These systems could include widely spaced single or multi-species tree, grass, and/or shrub rows with agronomic crops or pasture grass grown in the alleys (Table 1). The selection of companion perennial vegetation depends on landowner objectives and site suitability for a particular species. Expected benefits include improvements in environmental quality, economic returns, C sequestration, and wildlife benefits. In these systems, spatial heterogeneity exists in C stocks and sequestration due to tree and crop row configuration, differences in C input into the soil, decomposition rate, previous management, and associated soil micro fauna (Udawatta et al. 2008, 2009; Bambrick et al. 2010).

Only a few studies have examined above-ground biomass accumulation in alley cropping practices. In a 5 year-old alley cropping system in northeast Missouri (Udawatta et al. 2005), pin oak (*Quercus palustris* Muenchh) had twice the above-ground C of bur oak (*Quercus macrocarpa* Michx.) and swamp white oak (*Quercus bicolor* Willd.) (Table 2). The system sequestered 0.05 Mg C ha^{-1} in 5 years. The lower biomass accumulation of the site was attributed to persistent deer browsing during the initial 3 years of the study. Another study in Georgia, with *Albizia julibrissin* (L.) Benth. (mimosa) and grain sorghum (*Sorghum bicolor* (L.) Moench) during summer and wheat (*Triticum aestivum* L.) grown over winter, reported 50 times greater C than the Missouri study (Rhoades et al. 1998). The estimated tree density was 2,400 ha^{-1} (0.5 m spacing within rows and 4 m spacing between rows). The C input from pruning of leaves and twigs (second year at 1 m height) were 1.42 and 1.08 Mg ha^{-1} year^{-1}, respectively. In Southern Ontario, Canada, Peichl et al. (2006) showed that 13 year-old poplar and spruce alley cropping, and barley monocrop contained 96.5, 75.3, and 68.5 Mg C ha^{-1} (Table 2). In central Missouri, Pallardy et al. (2003) reported a biomass accumulation of 2.7 and 13 Mg ha^{-1} for first and second year harvests of poplar clones (1.3 and 6.5 Mg C ha^{-1}, assuming 50% C in the biomass).

Based on the limited data we estimate the above-ground C stock in alley cropping system as 26.8 Mg C ha^{-1} (Table 3). This is 4.6 times lower compared to the C stocks of riparian buffers. It should be noted that the alley cropping systems reviewed in this analysis are much younger (1–13 year-old) compared to the riparian buffers (2–250 year-old). We estimate that alley cropping has an average above-ground C sequestration potential of 2.7 Mg ha^{-1}.

In general 40–50% of C sequestered by trees is believed to be below-ground (Turnock 2001). However, this value changes by species and location because height growth, assimilation rates, litterfall, and root turnover differ by species. In an alley cropping practice in southern Ontario, Norway spruce (*Picea abies* L.) sequestered twice as much C as poplar in a 13 year-old study (Peichl et al. 2006). Although the above-ground C stocks of poplars and spruce were almost the same (85% and 82%, respectively), spruce had 63% of the C in branches and needles that provided greater quantities of litter material and thereby greater potential to add C to the soil pool. In addition, the spruce branches and needle were lignin-rich compared to poplar leaves and decomposed slowly in the soil.

In alley cropping, differences in SOC do not occur in a short period of time and in some situations, the SOC decreased with time (Thevathasan and Gordon 2004; Oelbermann et al. 2006a, b; Bambrick et al. 2010). According to Young (1997), tropical environments require at least 10 years to observe significant differences in SOC of alley cropping systems compared to monocropping systems. A longer timeframe is required to detect changes in the SOC content of these systems in the temperate zone due to colder climatic conditions and low C inputs (Peichl et al. 2006; Oelbermann et al. 2006a, b).

Studying 4, 8, and 21 year-old tree-based oat (*Avena sativa* L.)- maize-maize rotational alley cropping systems in Quebec, Canada, Bambrick et al. (2010) observed that differences in SOC among systems were not significant. However, spatial variation in SOC was obvious. The SOC concentrations were significantly greater at 0.75 m distance from the tree row than at 5 and 11 m. Also, 8 and 21 year-old sites showed significantly greater SOC concentration (77% and 12%, respectively) in the tree-based system than the conventional oat-maize rotations. The authors concluded that these systems required at least 6 years to sequester significantly more C in the soil than the conventional agricultural systems under existing soil and climatic conditions. Other studies, however, speculate that it would take at least 10 years to accrue significantly measurable differences in soil C between alley cropping and monocrop systems (Peichl et al. 2006; Oelbermann et al. 2006a, b).

The spatial variation in SOC in alley cropping systems results from the distinct spatial pattern of above-ground biomass and litterfall. Initially, more litter material tends to accumulate near the tree base (Bambrick et al. 2010). However, SOC concentration became non-significant with distance from trees as trees mature and spread roots and branches evenly. For example, Thevathasan and Gordon (2004) observed significant litter accumulation closer to the tree row and decreasing amounts away from the trees in a 6 year-old poplar-barley (*Hordeum vulgare* L.) alley cropping system in Ontario, Canada. The SOC content was 35% higher near the tree base and this effect extended up to 4 m in the crop alley when the system was 8 year-old. With time, crop alleys also showed increased SOC due to evenly distributed leaf biomass. The spatial variation in root biomass in alley cropping could also contribute to the SOC distribution. Jose et al. (2001) observed significantly greater root biomass in the black walnut (*Juglans nigra* L.) and red oak (*Quercus rubra* L.) tree rows compared to maize alleys in Indiana, indicating greater C stocks in the tree rows. Red oak root biomass was 2.1 and 1.8 times greater than the maize root biomass at the tree base and 1.1 m from the base. Black walnut had 1.1 and 1.37 times more roots, at those distances respectively, than maize. Trees had fewer roots at distances greater than 2.3 m from the tree row.

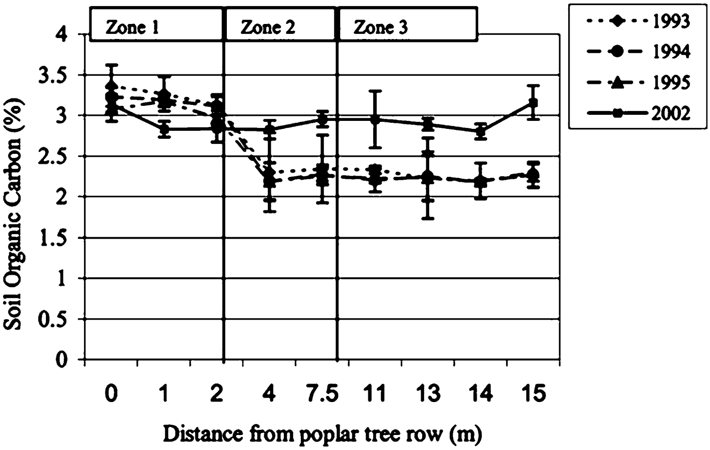

Fig. 5 Soil organic carbon concentrations in tree rows of 6- (1993), 7- (1994), 8- (1995), and 15- (2002)-year-old intercropped agroforestry practice in southern Ontario, Canada (Source: Thevathasan and Gordon 1997. Reproduced with permission)

Overall, soil C sequestration potential is much greater in alley cropping than in monocropping agronomic systems. For example, C inputs through litterfall on a poplar-spruce alley cropping with wheat-soybean-maize rotation were 0.6 and 0.95 Mg C ha^{-1} in the 11th and 12th years in Guelph, Ontario, Canada (Oelbermann 2002). In a 6 year-old hybrid poplar site (111 trees ha^{-1}) in Canada, Thevathasan and Gordon (1997) reported 1.07 Mg C ha^{-1} contributed by litterfall (Fig. 5). In the same study, hybrid poplar leaves and branches had C stocks of 1.3 and 5.5 Mg C ha^{-1} when trees were 13 year-old (Peichl et al. 2006). After 13 years the tree component of the system added 14 Mg C ha^{-1} in addition to the 25 Mg C ha^{-1} added by litter and fine roots (Thevathasan and Gordon 2004). The total C sequestration was therefore 39 Mg C ha^{-1}. The authors estimated that the system had immobilized 156 Mg ha^{-1} CO_2 or 43 Mg C ha^{-1} by age 13 and the system could potentially sequester significantly more C at the end of a 40 year harvest cycle.

One of the aspects neglected in soil C quantification in agroforestry is microbial C. In a pecan (*Carya illinoinensis* (Wangenh.) K. Koch)-cotton (*Gossypium hirsutum* L.) alley cropping system in Florida, Lee and Jose (2003) found significantly greater microbial biomass in a 47 year-old system compared to a 3 year-old system. Soils in the mature pecan system had 1.75 Mg C ha^{-1} (398 mg C kg^{-1} soil) as opposed to 0.38 Mg C ha^{-1} (88 mg C kg^{-1}) in the 3 year-old system (bulk density was assumed to be 1.25 g cm^{-3}). The highest SOM (4.3%) was also observed in the older alley cropping system and the authors attributed these differences to roots, leaves, branches, and other components from older pecan trees, as well as accrued, decomposing litter.

According to the USDA NASS (2008) and USDA NRCS (2003), cropland in the US is about 179 million ha, which includes approximately 16 million ha of idle land.

Montagnini and Nair (2004) and Nair and Nair (2003) estimated that approximately 80 million ha of land is available for alley cropping in the US and this represents 44.7% of the total cropland acreage. Garrett et al (2009) suggested that 40 million ha of highly erodible nonfederal croplands could be suitable for alley cropping. This represents 22% of the total croplands. Although alley cropping has the potential to sequester greater C compared to conventional agricultural practices, adoption of alley cropping has been slow in the US. We estimate that less than 10% of the croplands will be used for alley cropping in the near future. Using a 3.4 Mg C ha^{-1} year^{-1} C sequestration potential on 10% of the cropland (17.9 million ha), alley cropping practices in the United States could sequester 60.9 Tg C year^{-1}. If 80 million ha of cropland, as estimated by Nair and Nair (2003), is put under alley cropping, it would significantly increase the C sequestration potential to 272 Tg C year^{-1}.

According to Lal et al. (1999), 154 million ha of US cropland could sequester 73.8 Tg C year^{-1}. Another estimate by Nair and Nair (2003) shows that the 80 million ha of erodible and marginal agricultural land available for alley cropping in the US could potentially sequester 73.8 Tg C year^{-1} in above- and below-ground biomass. The C sequestration potential for US cropland and alley cropping, if expressed per ha, would be 0.479 (Lal et al. 1999) and 0.922 (Nair and Nair 2003) Mg C ha^{-1} year^{-1}, respectively. These estimates fall within the range (0.1–1 Mg C ha^{-1} year^{-1}) reported for improved agricultural management practices without incorporating perennial vegetation such as grasses, trees, and shrubs on cropland (CAST 2004). Our estimated C sequestration potential for alley cropping (3.4 Mg C ha^{-1} year^{-1}) is 7 times and 3.6 times greater than the estimates of Lal et al. (1999) and Nair and Nair (2003), respectively. However, the higher estimate is reasonable with the incorporation of trees as illustrated in Tables 2 and 3.

Silvopastoral Systems

Silvopasture is an agroforestry practice that intentionally integrates trees, forage crops, and livestock into a structural and functional system for optimization of benefits from planned biophysical interactions (Table 1). It is the most common form of agroforestry in North America (Clason and Sharrow 2000; Nair et al. 2008; Sharrow et al. 2009). In the US, approximately one-fifth of the forests or 54 million ha are grazed by livestock (Lubowski et al. 2006; Sharrow et al. 2009). In many regions, grazing also occurs either on marginal lands or as a secondary activity on high yielding timber lands. In temperate North America, silvopastoral systems have a great potential to sequester C due not only to high biological productivity, but also to the availability of larger areas under grazing management (Haile et al. 2008; Sharrow et al. 2009).

Conversion of pastureland to silvopasture has the potential to enhance rooting depth and distribution, quantity, and quality of organic matter input and thereby C sequestration potential (Haile et al. 2008). These systems could outperform C sequestration of either forest or pastures as they have both forest and grassland

Table 5 Soil organic carbon and nitrogen for the grazed pasture (GP), agroforestry buffer (AgB), grass buffer (GB), and row crop (RC) management treatments in Missouri, USA

Treatment	Soil organic carbon (% mass basis)	Total soil nitrogen (% mass basis)
GP	1.8[a]	0.20[a]
AgB	1.7[a]	0.20[a]
GB	1.7[a]	0.19[a]
RC	1.2[b]	0.13[b]

Source: Paudel et al. (2011). Reproduced with permission

Values with the same superscript within a column are not significantly different at $p \leq 0.05$

Table 6 Root dry weights and carbon to a 1-m soil depth in agroforestry (trees + grass; AgB), grass buffer (grass only; GB), rotationally grazed (RG), and continuously grazed (CG) treatments in a silvopasture practice in Missouri, USA

Treatment	Root dry weight (g 100 cm^{-3} soil)	C (g 100 cm^{-3} soil)
AgB	0.381	0.19
GB	0.475	0.23
RG	0.140	0.07
CG	0.074	0.04

Source: Kumar et al. (2010). Reproduced with permission

mechanisms of C capture that can maximize C sequestration both above- and below-ground. In general, trees store about 50–60% of the C in the above-ground biomass whereas pasture grasses store only 10% above-ground, the rest being allocated below-ground (Houghton and Hackler 2000; Sharrow and Ismail 2004). The greater potential to sequester C by silvopasture was illustrated by Sharrow and Ismail (2004) in their comparison of Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco)-cool season grass silvopastoral system with pasture and Douglas fir plantation in Oregon. These authors observed that the silvopastoral system sequestered an additional 0.74 Mg C ha^{-1} year^{-1} and 0.52 Mg C ha^{-1} year^{-1} than the plantation and pasture, respectively (Table 2). Individual trees in the silvopastoral systems grew faster than in conventional forests on the same site, allowing silvopastoral trees to store more C. The total amount of C stored in above- and below-ground biomass and soil was 5.8 and 8.2 Mg C ha^{-1} greater in silvopasture than pasture and Douglas fir plantation.

Roots of the perennial vegetation in silvopastoral systems shifts C deeper into the soil profile, compared to conventional pastures or row crops. Paudel et al. (2011) observed significantly greater percentages of C in soils under a cottonwood (*P. deltoides* Bortr. ex Marsh.) and grass silvopasture compared to maize-soybean rotation in Missouri (Table 5). In the same study area, Kumar et al. (2010) observed significantly greater root mass in the 1 m soil profile in tree-grass areas than the pasture grass (Table 6), clearly indicating the potential to deposit C deeper in the soil profile in silvopasture compared to pastures.

The spatial distribution of C, both above- and below-ground, can vary depending on the design of the silvopastoral systems and management practices. Soil organic

C derived from the tree component was significantly greater near the trees in a slash pine (*Pinus elliottii* Englem) and bahiagrass (*Paspalum notatum* Fluegge) silvopasture compared to open pasture areas in Florida (Haile et al. 2010). SOC contents were 1,033, 1,376, and 1,318 Mg ha^{-1} to a 1.25 m depth in open pasture, center of the pasture alley, and in-between trees in tree row, respectively (Table 2). Only the surface soil had more C derived from grasses. The SOC concentrations in open pastures were 94 and 26 Mg ha^{-1} for 0–75 and 75–125 cm depths, respectively. The silvopastoral system had 556 and 105 Mg ha^{-1} of SOC for the same depths, indicating the contribution of tree roots to the SOC pool, not only in the upper soil but also in the deeper soil profile.

Another factor that is not accounted for in many studies of silvopastoral systems is the amount of grass consumed by the grazing animals and the C deposited on soil via manure deposits. For example, sheep consumed a total of 30.5 Mg ha^{-1} forage in pastures and 22 Mg ha^{-1} of forage in silvopasture and deposited 9 and 7 Mg ha^{-1} manure in those two respective systems in the previously cited study in Oregon (Sharrow and Ismail 2004).

Strategies to enhance C sequestration in silvopaosture may include selection of complementary tree, shrub, and pasture grasses with optimal biomass accrual, deep rooting habits, and greater below-ground C accumulation potential. Proper maintenance of stocking rate, rotational grazing, and fertilizer application may also help enhance C sequestration. For example, Lee and Dodson (1996) estimated that conversion of 3.6 million ha marginal pasture lands in south central United States to silvopasture with pines could sequester 5.6 Tg C year^{-1} for the first 25 years and 1.1 Tg C year^{-1} for the subsequent 25 years. If this land is left for pasture, the sequestration would be 0.3 Tg C year^{-1}.

Although silvopasture remains the most common form of agroforestry in temperate North America, the precise land area under silvopasture is still unknown. Nair and Nair (2003) estimated the land available for silvopasture as 70 million ha. Pasture and grazed forestland areas in the United States are 237 and 54 million ha (www.ers.usda.gov/Data/MajorLandUses, accessed 24 December 2010), respectively. These land areas could be intensively managed for additional C sequestration.

The amount of SOC accrual in pasture lands ranged from 0.07 to 1.4 Mg C ha^{-1} year^{-1} (Franzluebbers 2005; Derner and Schuman 2007). According to Nair and Nair (2003), the C sequestration potential of silvopasture varies from a low of 1.8, medium 2.3, to a high of 3.3 Mg C yr^{-1}. Based on the data presented in the above sections, silvopastoral systems appear to sequester 6.1 Mg C ha^{-1} year^{-1} (Table 3). Using a sequestration potential of 6.1 Mg C ha^{-1} year^{-1} on 10% marginal pasture land (23.7 million ha) and 54 million ha of forests, the total C sequestration potential for silvopastoral lands in the Unites States could be as high as 474 Tg C year^{-1}. According to Montagnini and Nair (2004) and Nair and Nair (2003), 70 million ha of silvopasture in the US could store 9 Tg C year^{-1}. The value estimated in this analysis is 53 times greater than the previous estimate. We have used nearly the same acreage (77.7 million ha), but a much higher sequestration rate based on our literature review.

Windbreak

Windbreaks are designed with one or more rows of trees or shrubs planted across crop or grazing areas to reduce wind speed and enhance microclimate for crop and/or animal production (Table 1). Windbreaks have been used throughout history to protect homes, structures, livestock, and crops, control wind erosion and blowing snow, provide habitat for wildlife, improve landscape, and for odor mitigation (Brandle et al. 2004, 2009). Windbreaks are also used to reduce evaporation loss of water from soil and leaf surfaces (Brandle et al. 2009). The groundcover under the windbreak may also help reduce wind erosion and soil detachment by rain drops.

Like other agroforestry practices, windbreaks also offer great promise for C sequestration (Schoeneberger 2009). In addition to C sequestered by trees, windbreaks provide additional C sequestration due to improved crop and livestock production and energy savings (Kort and Turnock 1999). Indirectly, windbreaks reduce fuel use for heating and thereby reduce CO_2 emissions. Although shelterbelts and windbreaks have been planted in the Great Plains of the US since the 1930s, C sequestration in these systems have not been evaluated and there is a need for such estimates to determine the C sequestration capacity of these systems (Sauer et al. 2007).

The limited literature demonstrates the importance of species selection in maximizing the C sequestration potential of windbreaks or shelterbelts (Table 7). For example, hybrid poplar sequestered 367 kg C tree^{-1} in above- and below-ground compared to 110 kg C tree^{-1} in green ash (Kort and Turnock 1999). The above-ground C storage by single row conifer, hardwood, and shrubs for a windbreak in Nebraska was 9.14, 5.41, and 0.68 t km^{-1}, respectively (Brandle et al. 1992).

Table 7 Above- and below-ground biomass and carbon for shelterbelt trees commonly used in Saskatchewan, Canada

Vegetation type		Above-ground biomass	Below-ground biomass	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
	Seabuckthorn	213.0	106	160

Source: Kort and Turnock (1999). Reproduced with permission

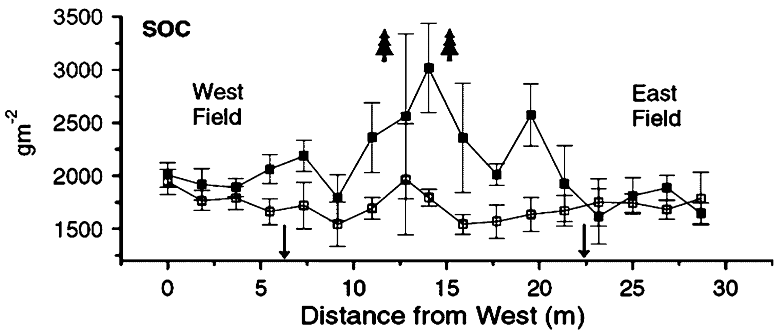

Fig. 6 Mean SOC values by position for 0–7.5 cm (closed squares) and 7.5–10 cm (open squares) across a shelterbelt crop transect in Nebraska, USA. Trees and arrows denote position of tree rows and extent of cultivation, respectively (Source: Sauer et al. 2007. Reproduced with permission)

In Saskatchewan, Canada, 17–90 year-old single row shelterbelts contained 24–41, 105, and 11 Mg C km^{-1} in conifer, poplar, and shrub shelterbelts, respectively (Kort and Turnock 1999). They also reported that above-ground C sequestration by hybrid poplar windbreaks in the US and Canada varied from <1 Mg C km^{-1} to >100 Mg C km^{-1}.

Windbreaks also contribute to the SOC pool albeit at a limited spatial scale on the landscape. In Nebraska, SOC concentration under a shelterbelt (3.04%) in the top soil (0–7.5 cm depth) was 55% more than that in the adjacent crop field (1.96%: Sauer et al. 2007). The shelterbelt treatment also contained 12% more SOC in the 7.5–15 cm depth compared to the crop field (Fig. 6). Overall, during a 35 year period, soils of 0–15 cm depth contained 3.71 Mg more SOC ha^{-1} in the shelterbelt than the cultivated region, which represents an annual sequestration of 0.11 Mg ha^{-1}. The authors attributed the increased SOC in the shelterbelt to absence of soil disturbance, increased inputs by litter, reduced erosion, and deposition of windblown material.

Nair and Nair (2003) estimated 85 million ha under windbreaks and sequestration potential of 4 Tg C year^{-1}. According to Brandle et al. (1992), 94 million ha of cropland in the North Central region need windbreaks to reduce damages. Another set of windbreaks are required to protect homes and roads. If 5% of the cropland, 120 million trees for protection of farmstead, and two million conifers for road protection are planted, these three categories would sequester 215, 13, 0.175 Tg C within 20 years or 11.4 Tg C year^{-1} (Brandle et al. 1992).

Based on C stocks in individual trees (Table 7) and considering a 20 year harvest cycle for 120 million hybrid polar trees and two million white spruce trees, windbreaks could potentially sequester 2.2 Tg C year^{-1} and 0.02 Tg C year^{-1}, respectively. The 5% (8.95 million ha) cropland with hybrid poplar could potentially sequester 131 Tg C or 6.56 Tg C year^{-1}. In this calculation, we considered 40 hybrid poplar trees for two rows of windbreaks per ha and 367 kg C tree^{-1} in a 20 year harvest cycle. Thus, the total C sequestration potential estimated for windbreaks is 8.79 Tg C year^{-1}.

Limitations, Implications, and Future Directions

Although above-ground biomass data are available for many tree and shrub species for forest stands, the literature lacks such information for integrated agroforestry systems. Since agroforestry trees are often open grown or grown in linear configurations, the growth patterns and hence C sequestration potential could vary from conventional plantations or forest stands. There is a need for data on above- and below-ground biomass and C for trees and shrubs under agroforestry practices for all regions. Specifically, such data are needed for stems, branches, bark, leaves, litter, nuts, roots and any material that is not removed from the site in order to estimate accurate C sequestration potential of agroforestry practices. Below-ground data such as root biomass, dynamics, and morphology are an integral part C sequestration in agroforestry. Soil C data are currently available mostly for the upper 10–35 cm soil profile. Some additional parameters such as bulk density, moisture %, rock volume %, and actual sampling depth are required to express C concentration and stock. Quantitative information on CO_2 and methane emission may provide data to refine estimates of net C sequestration. Sampling intensity, time, and age at which samples were collected affect the final estimate and such information should be included in the data sets as well.

Standardized experimental procedures and data gathering protocols for all regions are required so that data can be compared among regions. This also permits development of widely acceptable conclusions for larger geographic areas. Remote sensing and satellite data need to be used to accurately estimate C stocks and sequestration by agroforestry practices at larger spatial and temporal scales.

Trees and shrubs sequester C over longer periods than annual crops. In general, harvest cycle vary from 10 to 80 years for tree species commonly used in agroforestry systems. Research focus needs to be changed to understand long-term benefits of these multi-species systems. Since agroforestry practices with trees take two to three decades to mature, tree growth models under agroforestry practices are needed to estimate C sequestration. Complex models for tree growth with crop, pasture, and/or livestock may be simulated to understand long-term benefits and also to scale-up for larger regions. Models need data for initial calibration and validation and therefore research plots are required for all regions before models are simulated and specific conclusions are drawn regarding long-term effects. As explained earlier, tree growth or biomass equations for open grown agroforestry tree species need to be developed so that biomass and C can be estimated non-destructively.

Major statistical inventory systems (USDA Forest Service and NRCS) do not collect agroforestry statistics (Morgan et al. 2010). Therefore, updated and representative statistics are not available for agroforestry practices. A national inventory system may be developed to collect agroforestry statistics, including land area under specific practices.

Data should be used to develop agroforestry design criteria for all regions and practices that optimize C sequestration, environmental benefits, and economic returns. Agroforestry designs should include perennial vegetation with desirable characteristics such as greater C sequestration, greater below-ground C allocation, and other complementary effects for optimal C accrual. Intensive and improved management techniques may be implemented in concert with genetically improved species for fast growth and greater resource use efficiency (e.g. higher fertilizer use efficiency). Agroforestry practices with perennial vegetation could be designed to protect and enhance C sequestration on sensitive landscape locations such as highly vulnerable areas for nonpoint source losses and steep slopes. Improved agroforestry designs that are strategically placed on agricultural landscapes will eventually allow development of suitable mitigation strategies to enhance C sequestration.

Conclusions

There are several limitations in the data sets used for this analysis. Lack of accurate estimates of C sequestration for all regions and systems and land area under each agroforestry practice can introduce errors in the calculations. However, our estimate clearly indicates possible net gains in C sequestration that could be used to promote agroforestry as a promising CO_2 mitigation strategy in the US and potentially in other parts of North America. There are four main land use categories that can be considered as the most suitable for agroforestry in North America: degraded or non productive land, permanent agriculture and pasture land, forest land, and disconnected narrow riparian corridors. As the literature reveals, incorporation of agroforestry by introducing improved plant stock and implementing improved and intensive management techniques, C sequestration could be enhanced on this land base in a short period of time.

Since agroforestry was not inventoried by the major natural resources inventories, our estimates of C sequestration were based on several assumptions. A coarse approximation was made with limited data by multiplying the C sequestration in each system by the land area. A 4.7 Tg C year^{-1} C sequestration potential for riparian buffers was based on a 30 m wide buffer along both sides of 5% of total river length that would occupy 1.69 million ha. The estimated area was multiplied by 2.6 Mg C ha^{-1} year^{-1} accrual rate. The estimated potential value could be much higher if we had the buffer data for all water bodies. For alley cropping, we used 10% of the cropland and sequestration value of 3.4 Mg C ha^{-1} year^{-1}. The cropland in the US has the potential to sequester 60.9 Tg C year^{-1} through alley cropping. Using a sequestration potential of 6.1 Mg C ha^{-1} year^{-1} on 10% pasture land (23.7 million ha) and 54 million ha of forests, the total C sequestration potential for silvopastoral lands in the US could be as high as 474 Tg C year^{-1}. Windbreaks that protect cropland, farmstead, and roads could sequester 8.79 Tg C year^{-1}. The total potential C sequestration by agroforestry in the US is therefore 548.4 Tg year^{-1}. This could offset the current US CO_2 emissions (1,600 Tg C year^{-1} from burning fossil fuel such as coal, oil, and gas) by 34%.

Finally, we draw the following conclusions: (1) Agroforestry is a promising practice to sequester C (548.4 Tg year^{-1} in the US alone) while providing numerous environmental, economical, and social benefits in temperate North America (2) Rigorous, long-term C sequestration research in all regions and all agroforestry practices is required to develop accurate estimates and to develop policies and guidelines to recommend agroforestry practices that satisfy landowner expectations, (3) A standardized protocol is required for sampling, sample analysis, and data handling so that all available C data can be used to simulate models to examine long-term effects and to scale-up for larger landscapes, (4) An inventory of agroforestry practices is essential not only to accurately estimate C sequestration potential, but to quantify the economic and environmental impact of agroforestry, and (5) Future research should focus on developing design criteria for appropriate configuration, species selection, and planting density for various agroforestry practices to optimize C sequestration.

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