Carbon Sequestration Potential of Agroforestry Practices in Temperate North America

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Agroforestry strategies to sequester carbon in temperate North America

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Abstract Information on carbon (C) sequestration potential of agroforestry practices (AP) is needed to develop economically beneficial and ecologically and environmentally sustainable agriculture management plans. The synthesis will provide a review of C sequestration opportunities for AP in temperate North America and the estimated C sequestration potential in the US. We estimated carbon sequestration potential for silvopasture, alley cropping, and windbreaks in the US as 464, 52.4, and 8.6 Tg C yr^{-1}, respectively. Riparian buffers could sequester an additional 4.7 Tg C yr^{-1} while protecting water quality. Thus, we estimate the potential for C sequestration under various AP in the US to be 530 Tg yr^{-1}. The C sequestered by AP could help offset current US emission rate of 1,600 Tg C yr^{-1} from burning fossil fuel (coal, oil, and gas) by 33 %. Several assumptions about the area under different AP in the US were used to estimate C sequestration potential: 76 million ha under silvopasture (25 million ha or 10 % of pasture land and 51 million ha of grazed forests), 15.4 million ha (10 % of total cropland) under alley cropping, and 1.69 million ha under riparian buffers. Despite data limitation and uncertainty of land area, these estimates indicate the important role agroforestry could play as a promising CO_2 mitigation strategy in the US and temperate North America. The analysis also emphasizes the need for long-term regional C sequestration research for all AP, standardized protocols for C quantification and monitoring, inventory of AP, models to understand long-term C sequestration, and site-specific agroforestry design criteria to optimize C sequestration.

Keywords Alley cropping · Belowground carbon · Riparian buffers · Silvopasture · Windbreaks

Introduction

Current atmospheric CO_2 concentration is 36 % greater than the pre-industrial period concentration of 280 ppm. The projected doubling of atmospheric CO_2 by the latter half of the twenty-first century raises concerns for everyone. Society is demanding viable short- and long-term strategies to reduce atmospheric CO_2. To address this problem, the United Nations Environmental Program (UNEP) and World Meteorological Organization (WMO) formed the Intergovernmental Panel on Climate Change (IPCC) in 1988. The IPCC approved the establishment of the United Nations Framework Convention on Climate Change (UNFCCC) at the Rio de Janeiro, Brazil meeting held

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Table 1 Five main agroforestry practices and their predominant regions in temperate North America

Practice	Predominant regions	Function
1. Silvopasture	West and Southeast US	Economic diversification Improve animal health Create wildlife habitat Fire protection Timber management
2. Alley cropping	Midwest US	Increase biodiversity Increase income
3. Windbreaks	Great plains	Protect crop, animal, and structures Enhance crop and animal production Control erosion Distribute snowfall Mitigate odor
4. Riparian and upland buffers	All regions	Ameliorate non-point source pollution, protect watersheds and streambanks
5. Forest farming	All regions	Diversify income

Temperate North America for this manuscript denotes continental United States and southern half of Canada (*Source* Gold and Garrett 2009)

in 1992. In 1997, UNFCCC adopted the Kyoto Protocol which established binding targets for reducing the greenhouse gases.

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). In agroforestry practices (AP), woody plants, trees/shrubs are deliberately integrated into agriculture and animal production systems for multiple benefits arising from optimized biophysical interactions among the components (Gold and Garrett 2009; Table 1). Agroforestry is an appealing opportunity for agricultural lands to sequester significant amounts of carbon besides environmental, financial, and aesthetic benefits (Schoeneberger 2009; Calfapietra et al. 2010; Udawatta and Jose 2011). Sustainably managed AP can retain C for centuries (Dixon 1995). Although the amount of C stored on a site is a balance between long-term fluxes, the net C gain depends on the C content of the previous system that the AP replaces (Schroeder 1994).

The enhanced C sequestration concept in agroforestry is based on efficient use of resources by the structurally and functionally more diverse and complex plant communities compared to sole crop or grass systems (Marquez et al. 1999; Sanchez 2000; Sharrow and Ismail 2004; Thevathasan and Gordon 2004; Steinbeiss et al. 2008). Perennial vegetation is more effective than annual vegetation as it allocates a higher percentage of C to belowground and often extends the growing season (Morgan et al. 2010), therefore enhancing C sequestration potential of agricultural systems even further (Lal et al. 1998, 1999; Watson et al. 2000; Oelbermann et al. 2006a; Jose 2009). In addition, C stored by trees could stay in soils or as wood-products for extended periods. Since C sequestration in woody biomass of trees is large, the amount of C sequestered per unit area is substantially higher in agroforestry compared to conventional row-crop agriculture (Schoeneberger 2009).

Agroforestry practices accumulate more C than forests or pastures alone because they have both forest and grassland sequestration and storage patterns (Schroeder 1994; Kort and Turnock 1999; Sharrow and Ismail 2004). For example, an alley cropping system with pine trees and pasture utilizes light energy efficiently at multiple canopy levels compared to a traditional pasture. The diverse species planted in AP often have different physiological needs for particular resources and possess different structural or functional means to obtain them (Jose et al. 2004). If one species differs in utilization of even one of the components, for example light saturation of C_3 versus C_4 plants, C sequestration will be enhanced.

Adoption of AP has a greater potential to increase C sequestration of predominantly agriculture dominated landscapes than traditional agriculture without changing the land use to forestry (Lee and Jose 2003; Morgan et al. 2010; Nair and Nair 2003; Schoeneberger 2009; Nair et al. 2009). Compared to forestry that often has a large aerial footprint, many AP sequester substantial amounts of C on a relatively small land base, leaving a large portion of the landscape still available for agriculture (Ruark et al. 2003; USDA NAC 2000; Schoeneberger 2009).

A complete picture of C distributions in AP in the North American Continent is lacking in the literature

(Udawatta and Jose 2011), thus restricting development of suitable mitigation strategies to enhance C sequestration through establishment of AP 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 AP due to their complex nature, differences among practices, climatic conditions, and site factors. The objectives of this manuscript are to (1) provide a review of C sequestration opportunities available under various AP in temperate North America, and (2) synthesize available data and estimate C sequestration potential by AP in the US.

Data collection and analysis

A literature search was conducted to identify peer re-viewed papers and government documents related to C sequestration in the five major temperate AP namely; silvopasture, alley cropping, windbreaks, riparian buffers, and forest farming (Table 2). Scientific conclusions on C sequestration and storage as influenced by management practice, soil/site (type, restrictive horizons), precipitation (distribution, intensity, and quantity), and hydrology (slope and geology) were included in the analyses. Studies on C sequestration were categorized by practice. Forest farming was not included in the final analysis since sufficient information was not available for an in-depth review. When C concentration data were not provided, biomass was assumed 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 AP in the US only.

Silvopastoral systems

Silvopasture is the most common form of AP in North America with approximately one-fifth of the forests or 54 million ha grazed by livestock (Clason and Sharrow 2000; Lubowski et al. 2006; Nair et al. 2008; Sharrow et al. 2009). In temperate North America, silvopasture has great potential to sequester C due to high biological productivity, enhanced nutrient cycling, and the availability of larger areas under grazing management (Haile et al. 2008; Sharrow et al. 2009; Nyakatawa et al. 2011). Silvopasture could outperform C sequestration of both forests and pastures as they have both forest and grassland mechanisms of C capture that can improve total C sequestration both above and belowground. In general, trees store about 50–60 % of the C in the aboveground biomass whereas perennial pasture grasses store only 10 % aboveground, the rest being allocated to belowground (Houghton and Hackler 2000; Sharrow and Ismail 2004). Sharrow and Ismail (2004) reported that individual trees in their Douglas fir (*Pseudotsuga menziesii* [Mirb.] Franco)-cool season grass silvopasture in Oregon grew faster than in conventional forests on the same site, allowing silvopastoral trees to store more C (Table 3). The total amount of C stored (biomass and soil) was 5.8 and 8.2 Mg C ha^{-1} greater in silvopasture than pasture or Douglas fir plantation. Another study in the US Southeast with a goat (*Carpa aegagrus hircus*)-loblolly pine (*Pinus taeda* L.) silvopasture showed long-term environmental and economic benefits due to improved nutrient cycling and C sequestration (Nyakatawa et al. 2011).

The spatial distribution of C, both above and belowground is determined by the silvopastural design and its management (Haile et al. 2008, 2010; Paudel et al. 2011). Two studies in Missouri showed that soil organic carbon (SOC) and root mass were significantly greater in soils under a cottonwood (*P. deltoides* Bortr. ex Marsh.) silvopasture in the 1-m soil profile compared to maize (*Zea mays* L.)-soybean (*Glycine max* [L.] Merr) rotation (Fig. 1; Kumar et al. 2010; Paudel et al. 2011). Another study in Florida showed that SOC derived from trees was significantly greater throughout the 1.25-m soil profile and clay + silt fractions in a slash pine (*Pinus elliottii* Englem) and bahiagrass (*Paspalum notatum* Fluegge) silvopasture compared to open pastures (OP; Haile et al. 2010). SOC contents were 512, 618, and 565 Mg ha^{-1} in OP, center of the silvopastoral alley, and in the tree row, respectively (Table 4). The SOC content in OP were 450 and 62 Mg ha^{-1} for 0–50 and 50–125 cm depths, respectively, compared to corresponding values of 491 and 101 Mg ha^{-1}, respectively, in the silvopasture. While tree roots shift C deeper into the soil profile, diverse soil microbial communities that transform root C to SOC also help sequester more C through soil-atmospheric gas exchange (Haile et al. 2008, 2010; Nyakatawa et al. 2011).

Another factor that is often neglected in many silvopastoral studies is the amount of grass consumed

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

Agroforestry		Age	Species/treatment	Above or belowground^b	C (Mg ha^{-1})^a		Source
Practice	Location	(years)			Soil	Microbial	
Silvopasture	Oregon, USA		Pastures	0			Sharrow and Ismail (2004)
			Agroforestry	12.24			
			Plantation	6.95			
	Oregon, USA		*Understory C*				Sharrow and Ismail (2004)
			Pastures	1.0			
			Agroforestry	1.17			
			Plantation	2.23			
	Oregon, USA		Pastures		102.5		Sharrow and Ismail (2004)
			Agroforestry		95.9		
			Plantation		91.94		
	Florida, USA		Pasture		512		Haile et al. (2010)
			Center of alley between		618		
			tree row		565		
Alley cropping	Georgia, USA	1	Mimosa tree mulch with grain sorghum and winter wheat	2.5			Rhoades et al. (1998)
	Guelph, Ontario, Canada	15	Poplar intercrop	96.5			Peichl et al. (2006)
			Spruce intercrop	75.3			
			Barley sole crop	68.5			
	Missouri, USA	5	Pin oak	0.03			Udawatta et al. (2005)
			Bur oak	0.01			
			Swamp white oak	0.015			
	St. Remi, Quebec, Canada	8	Tree-based		77.1		Bambrick et al. (2010)
			conventional systems		43.5		
	Guelph, Ontario, Canada	21	Poplar		57		Bambrick et al. (2010)
			Norway spruce		51		
			conventional systems		51		
	Guelph, Ontario, Canada	15	Poplar intercrop		3.0		Peichl et al. (2006)
			Spruce intercrop		2.5		
			Barley sole crop		2.4 %		
	Florida, USA	3	Pecan orchard			1.2 %	Lee and Jose (2003)
		3	Pecan-cotton			1.9 %	
		47	Pecan orchard			4.3 %	
		47	Pecan-cotton			3.4 %	
	Florida, USA	3	Pecan system			0.38	Lee and Jose (2003)
		47	Pecan system			0.78	
Windbreak	Nebraska, USA	35	Windbreak		39.94		Sauer et al. (2007)
			Crop field		36.23		
Riparian buffers (aboveground)	Washington, USA	$\sim$250	N/A	9–271			Bailian and Naiman (2005)

Table 2 continued

Agroforestry		Age	Species/treatment	Above or belowground[b]	C (Mg ha^{-1})[a]		Source
Practice	Location	(years)			Soil	Microbial	
	Iowa, USA		Poplar	20			Tufekcioglu et al. (2003)
			Switchgrass	5			
			Cool season grasses	2.5			
	South Carolina, USA	2	N/A	<7.5			Giese et al. (2003)
		8		17.5			
		12		17.5			
		60		106			
	Northeast Ontario, Canada	95		29–269			Hazlett et al. (2005)
	Iowa, USA	6	Poplar-switchgrass-cool season grass	35			Tufekcioglu et al. (1999)
Riparian buffers (belowground)	Iowa, USA	6	Poplar	6			Tufekcioglu et al. (1999)
			Switchgrass	9			
			Cool season	7			
	South Carolina, USA	2	N/A	2.5			Giese et al. (2003)
		8		3.7			
		12		5			
		60		5.5			
	New York, USA			0.25–14.4			Kiley and Schneider (2005)
				mean 6.6			
	Iowa, USA	6	Poplar		2.4		Marquez et al. (1999)
			Switchgrass		1.8		
			Cool season grass		1.8		
			Crop (soybean)		0.4 Mg ha^{-1}yr^{-1}		
	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		Kim et al. (2010)
			Warm season grass		56.2		
			Cool season grass		57.8		
			Corn-soybean		57.1		

[a] Unless otherwise stated concentrations are in Mg ha^{-1}

[b] Assumed 50 % C in the biomass to estimate C when C concentration was not provided. Unless otherwise stated specified values represent aboveground C

by the grazing animals and the direct C deposition on soil of manure which contains about 60 % C (Franzluebbers and Doraiswamy 2007; Nyakatawa et al. 2011). For example, sheep (*Ovis aries*) consumed 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 (Sharrow and Ismail 2004).

Strategies that enhance C sequestration in silvopasture may include selection of complementary tree, shrub, and pasture grasses with optimal biomass accrual, deep rooting habits, and greater belowground

Table 3 Soil carbon content to a 125-cm soil depth in an open pasture and a silvopasture practice Florida, USA

Soil depth (cm)	Open pasture (Mg C ha^{-1})	Silvopasture alley	Silvopasture tree
0–50	450	517	465
50–75	41	35	55
75–125	21	66	45
0–125	512	618	565

Source Haile et al. (2008)

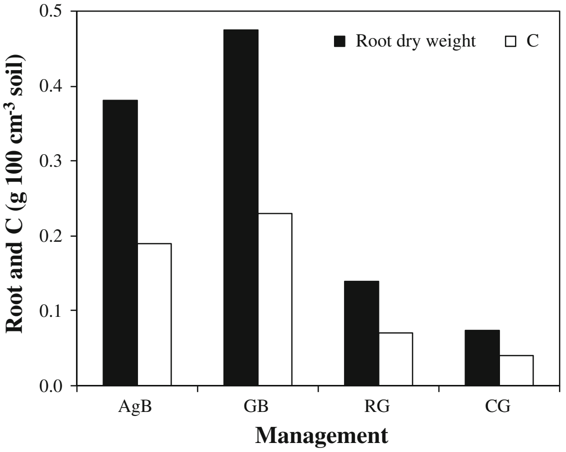


Fig. 1 Root dry weights and carbon to a 1 m soil depth in agroforestry (trees + grass; AgB), grass buffer (grass only; GB), rotationally grazed (RG), and continuously grazed (CG) treatments in a 8-year old silvopastural practice in Missouri, USA (Modified from Kumar et al. 2010)

C accumulation potential. Furthermore, proper stocking rate, rotational grazing, and fertilization can also enhance C sequestration.

Carbon sequestration potential of silvopasture can be estimated using the land area and sequestration rates. According to Nair and Nair (2003), the C sequestration potential of silvopasture varies from a low of 1.8 to a high of 3.3 Mg C yr^{-1}. Montagnini and Nair (2004) and Nair and Nair (2003) estimated that 70 million ha of silvopasture in the US could store 9 Tg C yr^{-1}. Lee and Dodson (1996) estimated that conversion of 3.6 million ha marginal pasture land in south-central US to silvopasture with pines could sequester 5.6 Tg C yr^{-1} for the first 25 years and 1.1 Tg C yr^{-1} for the following 25 years compared to 0.3 Tg C yr^{-1} for pasture. Based on 2007 statistics, pasture and grazed forests in the US are 248 and 51 million ha, respectively (USDA ERS 2011). Using a sequestration potential of 6.1 Mg C ha^{-1} yr^{-1} (Table 4) on 10 % marginal pasture land (25 million ha) and grazed forests (51 million ha), the total C sequestration potential for silvopastoral lands in the US could be as high as 464 Tg C yr^{-1}. Although we have used nearly the same acreage (76 million ha), our sequestration rate based on the literature review was much higher compared to the 1.8–3.3 Mg C yr^{-1} range of Nair and Nair (2003) resulting in 52 times greater sequestration than the previous estimate.

Alley cropping

Alley cropping includes widely-spaced rows with one or more species of trees and/or shrub in rows with agronomic crops grown in the alleys for improvements in environmental quality, microclimate, C sequestration, economic returns, and wildlife benefits (Table 1). The selection of companion perennial vegetation depends on landowner objectives and site suitability. In these systems, tree/shrub and crop row configuration, differences in C input into the soil, decomposition rate, previous management, and associated soil micro fauna determine C sequestration (Udawatta et al. 2008; Bambrick et al. 2010).

Rhoades et al. (1998) estimated C sequestration in an alley cropping system in Georgia with *Albizia julibrissin* trees planted at 0.5-m spacing within rows and an alley width of 4 m (tree density was 2,400 trees ha^{-1}) with grain sorghum (*Sorghum bicolor* (L.) Moench) during summer and wheat (*Triticale aestivum* L.) grown during winter. The C input from trees pruned at 1-m height in year two were 1.42 and 1.08 Mg ha^{-1} yr^{-1} from leaves and twigs, respectively (Table 2). The contribution from tree mulch was 2.5 Mg C ha^{-1}. In Southern Ontario, Canada, Peichl et al. (2006) showed that 13-year-old poplar alley cropping, spruce (*Picea abies* L.) alley cropping, and barley (*Hordeum vulgare* L.) monocrop contained 96.5, 75.3, and 68.5 Mg C ha^{-1}, respectively (Table 2). Another 5-year-old alley cropping study in Missouri, Udawatta et al. (2005) reported 0.024, 0.015, and 0.011 Mg ha^{-1} aboveground C for pin oak (*Quercus palustris* Muenchh), swamp white oak (*Q. bicolor* Willd.) and bur oak (*Q. macrocarpa* Michx.), respectively. The system had sequestered 0.05 Mg C ha^{-1} in 5 years (25 trees spp^{-1} ha^{-1}). The lower biomass accumulation of the site was

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

Practice	C stock[a]			C
	Minimum	Maximum	Mean	Sequestration rate[b]
	$\mathrm{Mg\ C\ ha^{-1}}$			$\mathrm{Mg\ ha^{-1}\ yr^{-1}}$
Silvopasture				
Aboveground	1.17	12.2	4.9	
Belowground				
Soil	1.03	1.38	1.21	
				6.1
Alley cropping				
Aboveground	0.05	96.5	26.8	
Belowground				
Soil	0.05	25	6.9	
				3.4
Windbreaks				
Aboveground	0.68	105		
Belowground				
Soil		23.1		6.4
			$\mathrm{kg\ tree^{-1}}$	$\mathrm{Mg\ ha^{-1}\ yr^{-1}}$
Hybrid poplar			367	0.73
White spruce			186	0.37
Riparian buffers				
Aboveground	7.5	269	123	
Belowground	2	14.4	4.6	
Soil	1.8	5.5	3.6	
				2.6

[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 belowground 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 $\mathrm{ha^{-1}}$ were assumed to estimate annual C accrual rates for windbreaks on cropland

attributed to persistent deer browsing during the initial 3 years of the study.

Carbon sequestration in alley cropping is also determined by plant material, duration, spatial variations in litter fall, and microbial contributions. Peichl et al. (2006) observed twice as much above and belowground C in poplar (15.1 t $\mathrm{ha^{-1}}$) than Norway spruce (6 t $\mathrm{ha^{-1}}$) in the tree portion of a 13-year-old study in southern Ontario. In alley cropping, differences in SOC do not occur in a short period of time and therefore, a longer timeframe is required to detect changes in the SOC content of these systems especially in the temperate zone due to colder climatic conditions and low C inputs as compared to tropical environments (Young 1997; Oelbermann 2002; Thevathasan and Gordon 2004; Peichl et al. 2006; Oelbermann et al. 2006a, 2006b; Bambrick et al. 2010). For example, in Quebec, Canada, Bambrick et al. (2010) observed no differences among 4-, 8-, and 21-year-old tree-based oat (*Avena sativa* L.)-maize–maize rotational alley cropping. However, the SOC differences were significantly greater at 0.75-m distance from the tree row than 5- and 11-m distances in a hybrid poplar system. The 8-year-old site had 77 % more SOC and the 21-year-old site had only 12 % more SOC than the conventional oat-maize rotations.

The spatial variation in SOC in alley cropping systems are caused by distinct spatial patterns of aboveground biomass, litterfall, and roots. 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 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. Another study in Guelph, Canada showed C inputs through litterfall on a poplar-spruce alley cropping with wheat-soybean-maize rotation were 0.6 and 0.95 Mg C $\mathrm{ha^{-1}}$ in the 11th and 12th years (Zhang 1999; Oelbermann 2002). With time, crop alleys also showed increased

SOC due to evenly distributed leaf biomass. In a 6-year-old hybrid poplar site (111 trees ha^{-1}) in Canada, Thevathasen and Gordon (Thevathasan and Gordon 1997) reported 1.07 Mg C ha^{-1} was contributed by litterfall. 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 trees (branches, stem, and leaves) 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). Jose et al. (2001) also observed significantly greater root biomass in the black walnut (*Juglans nigra* L.) and red oak (*Q. 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, respectively.

One of the aspects neglected in soil C quantification in AP is microbial C. In a pecan (*Carya illinoinensis*)-cotton (*Gossypium hirsutum*) 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} for the 35-cm sampling depth). The highest soil organic matter (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 deposited in previous years by the 47-year-old trees, compared to 3-year-old trees.

Based on 2007 statistics, cropland in the US is between 144 and 165 million ha (USDA NASS 2008; USDA NRCS 2009; USDA ERS 2011). Montagnini and Nair (2004) and Nair and Nair (2003) estimated that approximately 80 million ha of land is available for alley cropping and this represents 52 % of the total mean cropland acreage (154 million ha). Garrett et al. (2009) suggested that 40 million ha of highly erodible nonfederal cropland could be suitable for alley cropping or approximately 26 % of the total cropland. We estimate that less than 10 % of the crop land may be used for alley cropping in the near future. Using a 3.4 Mg C ha^{-1} yr^{-1} C sequestration potential on 10 % of the crop land (15.4 million ha), alley cropping practices in the US could sequester 52.4 Tg C yr^{-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 yr^{-1}.

Lal et al. (1999) estimated 100 Tg C yr^{-1} sequestration on 154 million ha of US cropland. Another estimate by Nair and Nair (2003) showed that alley cropping in the US could potentially sequester 73.8 Tg C yr^{-1} in above and belowground biomass. The C sequestration potential for US cropland and alley cropping, would be 0.65 (Lal et al. 1999) and 0.922 (Nair and Nair 2003) Mg C ha^{-1} yr^{-1}, respectively. These estimates fall within the range (0.1–1 Mg C ha^{-1} yr^{-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} yr^{-1}) is 5.2 times and 3.7 times greater than the estimates of Lal et al. (1999) and Nair and Nair (2003). However, the higher estimate is reasonable with the incorporation of trees as illustrated in Tables 2 and 3.

Windbreak

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, reduce evaporation loss of water from soil and leaf surfaces, and for odor mitigation (Table 1; Brandle et al. 2004, 2009). Windbreaks, also referred to as shelterbelts, are linear in shape and consist of one or more rows of trees or shrubs planted across crop or grazing areas to reduce the wind speed, enhance the microclimate, and mitigate odor. Although only a small portion of agricultural land is occupied by windbreaks, it can sequester significant amount of C per unit land area compared to many agricultural practices (USEPA 2006; Schoeneberger 2009). Windbreaks indirectly reduce CO$_2$ emissions through improved crop and livestock production and energy savings due to reduced fuel use for heating thereby help mitigate global warming (Kort and Turnock 1999). Although windbreaks have been planted in the Great Plains of the US since the 1930s, accurate estimates of C sequestration potential is missing in the literature, and there is a need for such estimates to determine the C sequestration capacity of these systems (Sauer et al. 2007).

Since C sequestration varies with tree species, age, and location, selection of site-soil-climate compatible species can optimize C sequestration (Figs. 2, 3; Brandle et al. 1992; Kort and Turnock 1999; Hou et al. 2011). For example, green ash (*Fraxinus pennsylvanica* Marsh.) sequesters 110 kg C tree^{-1} compared to 367 kg C tree^{-1} in hybrid poplar (Fig. 3; Kort and Turnock 1999). In Nebraska, soils under shelterbelts sequestered 0.11 Mg ha^{-1} during a 35-year study (Sauer et al. 2007). Hou et al. (2011) used site index, climate, and tree data to simulate biomass and C sequestration in Nebraska using a spatial model. They suggested that models could estimate C sequestration with minimum number measurements for a region (Fig. 2).

According to Brandle et al. (1992), establishing windbreaks on 85 million ha of unprotected cropland would sequester 59 million metric tons of C at age 20. Similarly, planting windbreaks around unprotected farms would sequester another 3.5 million metric tons of C within 20 years. If windbreaks were established on 5 % of the cropland, 120 million trees were planted for protection of farmsteads, and two million conifers were planted for road protection, the respective C sequestration would be 215, 13, and 0.175 Tg C within 20 years or 11.4 Tg C yr^{-1} for windbreaks in the US (Brandle et al. 1992). Nair and Nair (2003) estimated that windbreaks could sequester 4 Tg C yr^{-1} on 85 million ha of US farmlands. Based on C sequestration of trees from Fig. 3 and considering a 20-year harvest cycle for 120 million hybrid poplar trees and 2 million white spruce trees, windbreaks could potentially sequester 2.02 Tg C yr^{-1}. Cropland (5 % or 8.95 million ha) could sequester 6.56 Tg C yr^{-1} with hybrid poplar. Potential C sequestration by windbreaks is 8.58 Tg C yr^{-1}. Differences among estimates could be due to species, potential area under windbreaks, and the data used in these calculations.

Riparian buffers

Riparian buffers 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 (Welsh 1991; Lowrance et al. 1985).

In general, potential C sequestration and storage are greater in riparian buffer systems compared to row crops or upland forests. However, Naiman et al. (2005) caution that net primary production in riparian buffers is determined by species composition, planting density, age, soil, climate, management and spatial configuration of different functional groups, which affects biomass accumulation and C sequestration. According to Naiman et al. (2005) the aboveground C of a mature riparian forest ranges from 50 to 150 Mg ha^{-1}. As a riparian system matures, the above and belowground biomass of the understory and overstory vegetation increases, giving an overall increase in the system-level C stock within plants and soil. In their study, Naiman et al. (2005) observed that stem biomass accrual increased at a diminishing rate for stands >150-year and reached a plateau after 250-year (Fig. 4). Biomass accumulation pattern of another riparian system in Washington, USA, supports the above, 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 also observed results similar to Balian and Naiman (Bailian and Naiman 2005). In their study, aboveground C increased from 0.5 to 97 Mg ha^{-1} while stem density decreased from >10,000 to <1,300 stems ha^{-1} over the 60-year period in the willow (*Salix* spp.) and cottonwood riparian buffer. Another study in Iowa, Tufekcioglu et al. (2003) observed 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, respectively (Fig. 5). Giese et al. (2003) observed 106 Mg ha^{-1} C in a 60-year-old riparian buffer compared to <7.5 Mg ha^{-1} in 2-year-old, 17.5 Mg ha^{-1} in 8-year-old, and 17.5 Mg ha^{-1} in 12-year-old buffers 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. Estimates

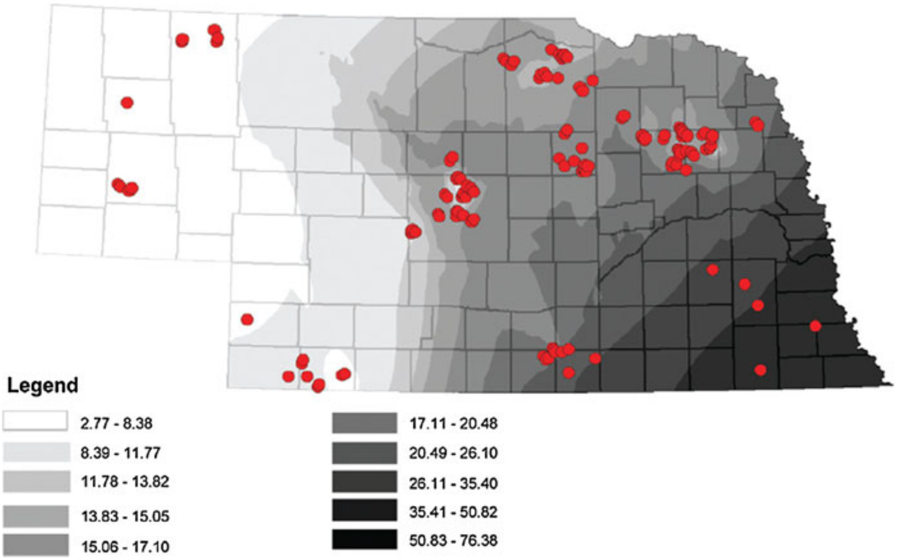
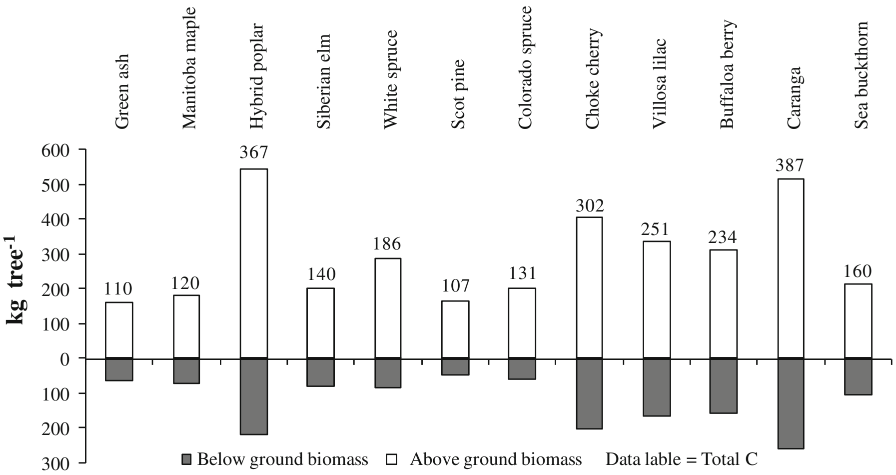


Fig. 2 Output of a spatial model predicting green ash aboveground volume per tree calibrated with actual age 36 in Nebraska. Sampling points for windbreaks in *red* (*Source* Hou et al. 2011). (Color figure online)

Fig. 3 Above and belowground biomass and total carbon content for shelterbelt trees commonly used in Saskatchewan, Canada (modified from Kort and Turnock 1999)



were from 29.3 to 269.1 Mg C ha^{-1} for riparian plots and 28.3 Mg to 135.1 Mg C ha^{-1} for upland plots (Table 2).

The aforementioned studies provide a realistic estimate of aboveground C stock of mature riparian buffer systems in temperate North America. We estimated an average aboveground C accumulation of 123 Mg C ha^{-1} for a mature riparian buffer with a 50-year harvest cycle. The estimated average aboveground C sequestration potential is 2.46 Mg C ha^{-1} yr^{-1} (Table 4). In Canada, C stocks of 29.3 to 269.1 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} aboveground storage for temperate zone riparian buffers with a 30-year harvest cycle. Our mean estimate of 123 Mg C ha^{-1} is twice the value estimated by Schroeder (1994). According to Hoover and Heath (2011), aboveground 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 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 belowground and this C is retained in the soil C pool as roots decay. Studying root densities in riparian-crop transects in Iowa,

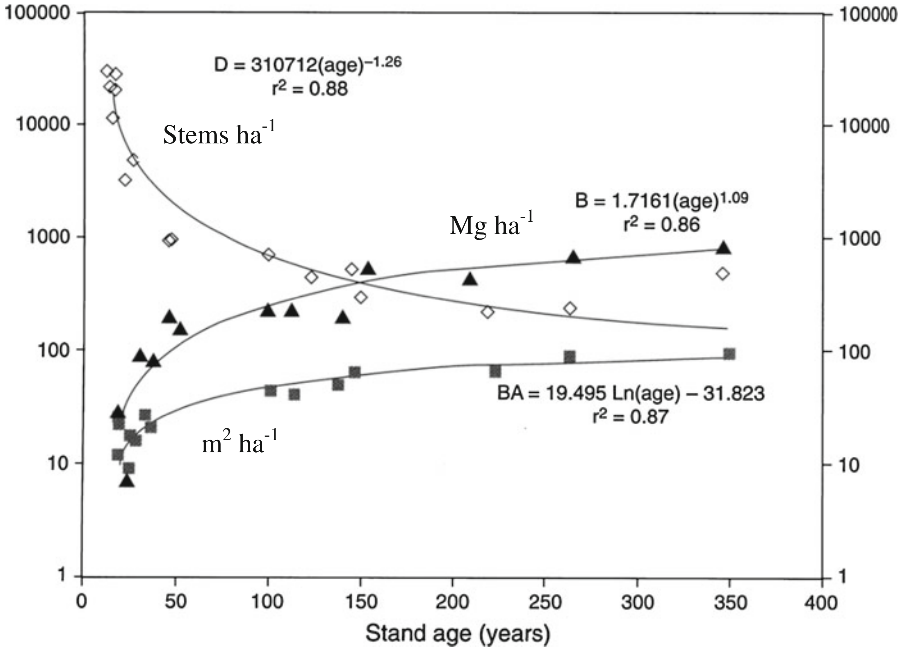


Fig. 4 Changes over time in stem density (D, stems ha^{-1}), biomass (B, Mg ha^{-1}), and basal area (BA, m^2 ha^{-1}) of a riparian forest buffer in Washington, USA. *Y* axis in logarithmic scale for stems ha^{-1}, Mg ha^{-1}, and m^2 ha^{-1} (*Source* Naiman et al. 2005)

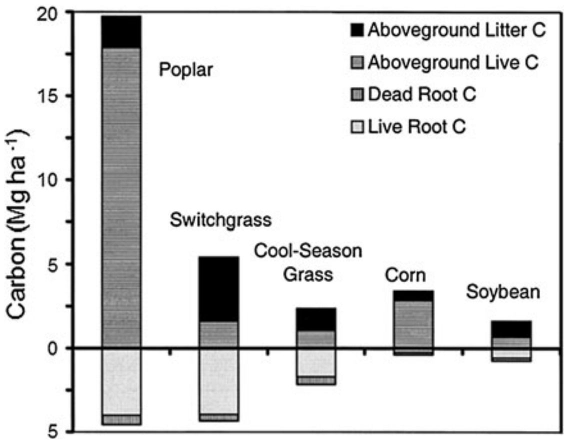


Fig. 5 Litter and root carbon distributions in a 5-year-old riparian system with trees, grass, and crops in Iowa, USA (*Source* Tufekcioglu et al. 2003)

Tufekcioglu et al. (1999) found greater root biomass in the riparian vegetation compared to the row-crop areas, suggesting that there was also greater C in the riparian areas. On average the root C for poplar (*P. deltoides* × *nigra* DN-177), switchgrass, and cool season grass were 3, 4.5, and 3.5 Mg C ha^{-1} (Table 2). The riparian vegetation consisting of trees and grasses also had more fine (0–2 mm dia.) and medium (2–5 mm dia.) roots in the surface soil and throughout the 165-cm soil profile. Coarse and medium roots of poplar trees extended beyond 165-cm depth while no crop roots were found below 125 cm. Four years since establishment, root biomass and, presumably belowground C, were significantly greater in the riparian zone than the row crop areas (Marquez et al. 1999).

Another study, also from Iowa, demonstrates the potential to sequester greater quantities of C in soils under riparian buffers compared to row-crops. Belowground C in the tree and switchgrass areas of the riparian buffers was significantly greater than in accompanying grasses and adjacent corn-soybean crop areas (Tufekcioglu et al. 2003). Belowground C was >4.5 Mg ha^{-1} for poplar and switchgrass areas of the riparian buffer compared to <2 Mg ha^{-1} for cool season grasses and <1 Mg ha^{-1} for corn and soybean (Fig. 5). Similar belowground C accrual rates were reported by Giese et al. (2003) in South Carolina for a mature riparian buffer (Fig. 6). The results showed belowground biomass of 5, 7.5, 10, and 11 Mg ha^{-1} in 2-, 8-, 12-, and 60-year-old riparian buffers (Fig. 6; Giese et al. 2003). Using the 50 % conversion rate, these buffers contained 2.5, 3.7, 5.0, and 5.5 Mg C ha^{-1}, respectively (Fig. 7). The study also showed that fine root biomass in the

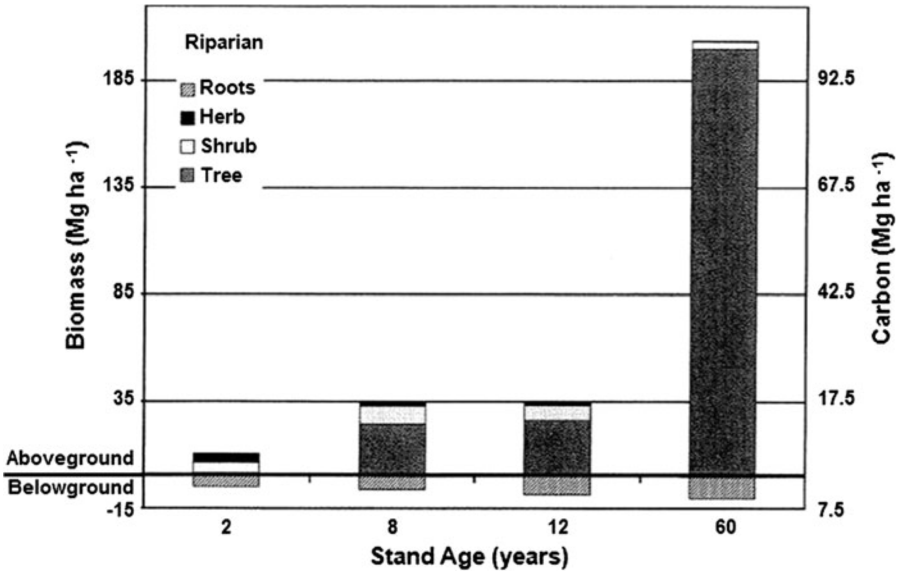


Fig. 6 Above and belowground biomass and carbon of 2-, 8-, 12-, and 60-year-old riparian stands in South Carolina, USA (*Source* Giese et al. 2003)

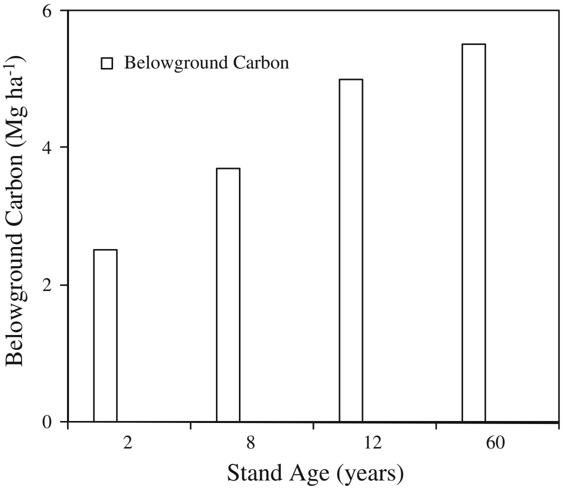


Fig. 7 Belowground carbon as 50 % of the biomass in 2-, 8-, 12- and 60-year-old riparian buffer system in South Carolina, USA (*Source* Giese et al. 2003, reproduced with permission)

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). Using the aforementioned results, we estimate a mean C sequestration of 0.09 Mg C ha^{-1} yr^{-1} in belowground tissues for riparian buffer systems for a 50-year harvest cycle (Table 3).

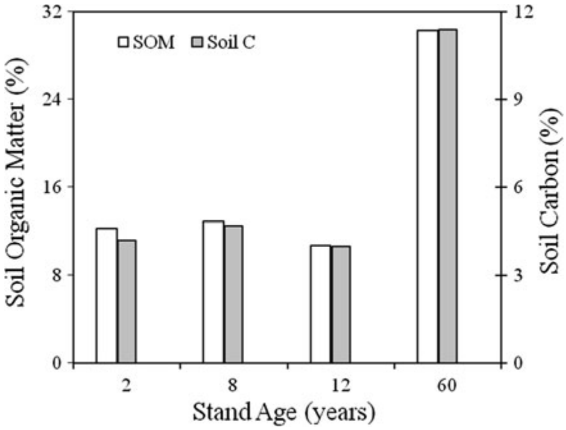


Fig. 8 Soil organic matter and soil carbon percentages in 2-, 8-, 12- and 60-year-old riparian buffer system in South Carolina, USA (*Source* Giese et al. 2003, reproduced with permission)

Marquez et al. (1999) and Giese et al. (2003) explained that soils under mature riparian stands compared to monocropped agroecosystems or younger riparian buffers contain significantly more SOM which contains about 50 % C. Results from a poplar—grass riparian system compared with adjacent crop areas in Central Iowa showed that changes in soil C can occur in a relatively short period of time (Marquez et al. 1999). In the poplar and switchgrass zones, SOC accrued at 1.2 and 0.9 Mg C ha^{-1} yr^{-1} rate and SOC were significantly greater than in the crop zone in 1st and 6th year samplings. From a mature riparian buffer in South Carolina, Giese et al. (2003) observed 2.6 times greater soil C content in the 60-year-old buffer compared to 2-, 8-, and 12-year-old riparian buffers (Fig. 8). Kim et al. (2010) compared soils to a 15-cm depth in riparian, warm season, and cool season grass buffersin Iowa and showed a SOC increase of 50–71 Mg ha^{-1} in 7 years in the riparian buffer while the grass buffers contained only 47 and 56 Mg C ha^{-1}, respectively.

The litter material in the riparian zones, either from the riparian vegetation or flooding, also contributes to soil C sequestration. The litter was approximately 47 % of the aboveground biomass production Amazonian floodplanes (Piedade et al. 2001). 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} yr^{-1} (Tufekcioglu et al. 2001). The study showed that the annual respiration rate was the highest in the streamside cool season grass buffer (12.2 Mg C ha^{-1} yr^{-1}) and the lowest in the corn areas (7.4 Mg C ha^{-1} yr^{-1}). Annual crop side respiration rates were 11.5, 11.4, 10.3, and 7.5 Mg C ha^{-1} yr^{-1} for cool season grass, poplar buffer, switchgrass, and soybean, respectively. The results of the study show that perennial vegetation in the buffer areas, either trees or grasses, had significantly greater respiration rates compared to the annual crops. However, trees leaf out and grasses begin to grow before annual crops are established and thereby begin to store C sooner in the above and belowground vegetation thus increasing the overall C sequestration potential of the system (Marquez et al. 1999).

As the above literature reveals, riparian buffer systems have tremendous potential to sequester C in above and belowground 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 highest during the early stages of development. Root C deposited in the deeper horizons of the soil profile could remain for extended periods, and thereby contribute 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.

State and national levels agencies recommend buffer widths between 15 and 100 m for protection of water bodies, water quality improvements, streambank stabilization, and to reduce sediment and nutrient losses (USDA NRCS 2007). 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 %. Buffers of 91-m width have been proposed for levee protection and reduce flood damage (Dwyer et al. 1997). The composition and the width of the riparian buffer system vary and much wider buffers or multi zone buffers are required for the removal of soluble nutrients compared to stabilizing streambanks (Schultz et al. 2009). Properly designed riparian buffers also improve 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 (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 of conservation buffers by 2002 (USDA NRCS 2002). A 30-m wide riparian buffer along both sides of 5 % of total river length would occupy 1.69 million ha. Using a conservative estimate of 2.6 Mg C ha^{-1} yr^{-1} accrual rate (Table 4), the potential C sequestration by riparian buffers along rivers in the US could be as high as 4.7 Tg C yr^{-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. The 4.7 Tg yr^{-1} C sequestration potential estimated by this analysis is significantly greater than the 1.5 Tg C yr^{-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

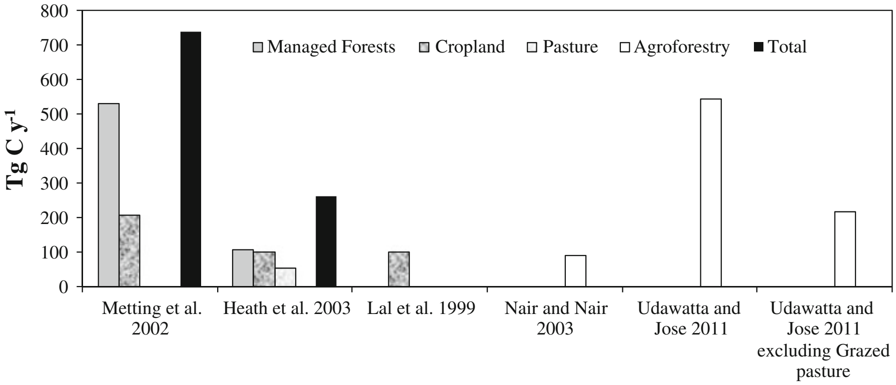


Fig. 9 A comparison of carbon sequestration potential for different management systems

30-m wide forested riparian buffer zone on one-fourth of the 3.2 million km of conservation buffers committed by the USDA in 2002 for their estimate.

Limitations, implications, and future directions

Our C sequestration estimates are within the range reported for various land use systems in the US (Fig. 9). There are different estimates for the same land use systems due to differences in estimation methods. For example, C sequestration potential can be as high as 737 Tg yr^{-1} for managed forests and cropland according to Metting et al. (2002). Heath et al. (2003) estimated a total potential of 260 Tg yr^{-1} for managed forests, cropland, and pasture. The sequestration potential estimated by Heath et al. was only 35 % of that reported by Metting et al. (2002). Nair and Nair (2003) estimated a C sequestration potential of 90 Tg yr^{-1} for agroforestry. It was only 16 % of the 545 Tg yr^{-1} potential estimated by Udawatta and Jose (2011). The Udawatta and Jose (2011) value is close to the sequestration of managed forest estimated by Metting et al. (2002). With new information used in this revised analysis, our estimated C sequestration potential by US agroforestry is 530 Tg yr^{-1}. If silvopasture on grazed forests are removed from this estimate, the C sequestration would only amount to 219 Tg yr^{-1}.

Although aboveground biomass data are available for many tree and shrub species on forestland, the literature lacks such information for integrated AP (Raeker et al. 2011). Since trees in AP are often open grown (less than 10 % stocking) or grown in linear configurations, the growth patterns and hence C sequestration potential are different from conventional plantations or forest stands. There is a need for data on above and belowground biomass and C storage for trees and shrubs under AP 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 AP.

Soil C data for 0–1 or 0–2 m soil depths and additional parameters such as bulk density, moisture %, rock volume %, and actual sampling depth are required to express C concentration and stock. Sampling intensity, time, and age at which samples are collected affect the final estimate and such information should be included in the data sets as well. Quantitative information on CO_2 and methane emission may provide data to refine estimates of net C sequestration.

Standardized experimental procedures and data gathering protocols for all land use systems and regions are required so that data can be compared among land uses and regions. Remote sensing and satellite data may be used with ground truthing to accurately estimate C stocks and sequestration by AP at larger spatial and temporal scales.

Since harvest cycles vary from 10 to 80 years for tree species, research focus should be changed to understand long-term benefits of multi-species and multi-year systems. Research also should address interactions among various components in AP, C contributions, and optimum conditions for maximum C sequestration. Simulation of tree growth models integrated with GIS, weather, soil, landscape, and management could help understand long-term benefits of AP and also to scale-up for larger regions. Models need data for initial calibration and validation as has been done in Nebraska (Fig. 2) and therefore research plots are required for other regions (of the US or temperate NA) before models are simulated and specific conclusions are drawn regarding long-term effects.

Forest and cropland inventory systems currently used by the USDA's Forest Service, National Agricultural Statistics Service, and Natural Resources Conservation Service do not collect agroforestry statistics; therefore, updated and representative statistics are not available for AP. 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 to optimize C sequestration, environmental benefits, and economic returns. When economic or other considerations are equal, agroforestry designs should include perennial vegetation with desirable characteristics such as greater C sequestration, greater belowground 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. Agroforestry practices with perennial vegetation could be designed to protect and enhance C sequestration on sensitive landscape locations such as with greater land degradation and water pollution potential. Improved agroforestry designs that are strategically placed on agricultural landscapes will eventually allow development of suitable mitigation strategies to enhance C sequestration.

Conclusion

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 AP 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. Incorporation of agroforestry by introducing improved planting stock and implementing improved and intensive management techniques, C sequestration could be enhanced on this land base in a short period of time.

Our estimates of C sequestration were based on several assumptions as agroforestry is not inventoried by major natural resources inventories. 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 yr^{-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} yr^{-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 conservatively estimated the potential conversion of 10 % of the crop land with a sequestration value of 3.4 Mg C ha^{-1} yr^{-1} to potentially sequester 52.4 Tg C yr^{-1} through alley cropping. Using a sequestration potential of 6.1 Mg C ha^{-1} yr^{-1} and conversion of 10 % of pasture land (25 million ha) and 51 million ha of forests to silvopastural practices, the total C sequestration potential for silvopasture in the US could be as high as 464 Tg C yr^{-1}. Establishment of windbreaks that protect cropland, farmstead, and roads could sequester an additional 8.6 Tg C yr^{-1}. The total potential C sequestration by agroforestry in the US is therefore 530 Tg yr^{-1}. This could offset the current US CO_2 emissions (1,600 Tg C yr^{-1} from burning fossil fuel such as coal, oil, and gas) by 33 %.

Finally, we draw the following five conclusions. (1) Agroforestry is a promising practice to sequester C (530 Tg yr^{-1} in the US alone) while providing numerous environmental, economical, and social benefits (2) Rigorous, long-term C sequestration research on site-management appropriate AP in all regions is required to develop accurate estimates and to develop policies and guidelines to recommend AP that satisfy landowner expectations, (3) A standardized protocol is required for sampling, sample analysis, and data handling so that C data can be used to simulate models to examine long-term effects and to scale-up for larger landscapes, (4) An inventory of AP 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 AP to optimize C sequestration.

Acknowledgments This manuscript is an improved and updated version of Udawatta and Jose (2011) cited in the references. We gratefully acknowledge Dr. Jerry Van Sambeek for his thoughtful comments and suggestions on an earlier version of this manuscript.

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