Chapter 3
Greenhouse Gas Mitigation and Accounting

Michele Schoeneberger and Grant Domke

Michele Schoeneberger is research program lead and soil scientist (retired), U.S. Department of Agriculture (USDA), Forest Service, USDA National Agroforestry Center; **Grant Domke** is team leader and research forester, USDA Forest Service, Forest Inventory and Analysis.

Contributors: P.K.R. Nair, Marlen Eve, Alan Franzluebbers, Christopher Woodall, Toral Patel-Weynand, James Brandle, and William Ballesteros Possu

P.K.Ramachandran Nair is the distinguished professor of agroforestry at the School of Forest Resources and Conservation, University of Florida; **Marlen Eve** is the national program leader for Natural Resources and Sustainable Agricultural Systems, USDA Agricultural Research Service; **Alan Franzluebbers** is a research plant physiologist, USDA Agricultural Research Service; **Christopher Woodall** is project leader and research forester, USDA Forest Service, Northern Research Station, Center for Research on Ecosystem Change; **Toral Patel-Weynand** is the director of Sustainable Forest Management Research, USDA Forest Service, Research and Development; **James Brandle** is emeritus shelterbelt professor, School of Natural Resources, University of Nebraska; **William Ballesteros Possu** is associate professor, agrarian sciences faculty, University of Nariño.

It is well documented that forests provide significant carbon (C) sequestration (McKinley et al. 2011). In the 2013 U.S. Department of Agriculture (USDA) Greenhouse Gas Inventory, forests, urban trees, and harvested wood accounted for the majority of agricultural and forest sinks of carbon dioxide (CO_2) (USDA-OCE 2016). Trees outside of forests (TOF; assemblages of trees not meeting the definition of forest based on area, width, and/or canopy coverage criteria) also play an important role in C sequestration as well as in the reduction of greenhouse emissions but do so within agricultural and urban lands. While the positive contributions of these forest-derived mitigation services within U.S. agricultural lands have been documented, the lack of inventory- and activity-specific data limits our ability to assess the amount and therefore significance of these contributions (Robertson and Mason 2016, Schoeneberger 2009). Recent international studies, however, indicate these contributions can be very significant in regards to overall global and national C budgets (Schnell et al. 2015, Zomer et al. 2016).

According to the most recent U.S. Agriculture and Forestry Greenhouse Gas Inventory (1990–2013), agriculture in the United States contributed 595 million metric tons of CO_2 equivalents in 2013, with nearly one-half (45 percent) of these emissions coming from soils, 28 percent from livestock production (enteric fermentation), and the rest from energy use and managed livestock waste (USDA-OCE 2016). Agriculture also has the ability to offset these emissions through the use of management practices, of which TOF-based practices and, specifically, the TOF practice of agroforestry, are now included (CAST 2011, Denef et al. 2011, Eagle and Olander 2012, Eve et al. 2014). Agroforestry is the intentional integration of woody plants into crop and livestock production systems to purposely create a number of forest-derived services that support agricultural operations and lands, including those services that can directly address greenhouse gas (GHG) mitigation and adaptation needs related to food security and natural resource protection under changing conditions (see chapter 2 in this assessment, Nair 2012a, Plieninger 2011, Schoeneberger et al. 2012, Verchot et al. 2007, Vira et al. 2015). It is because of this capacity to simultaneously provide C sequestration and other GHG mitigation services, along with adaptation, that interest is growing in the use of agroforestry and other TOF systems (e.g., woody draws, woodlots, fencelines) in U.S. agricultural climate change strategies (CAST 2011, Ogle et al. 2014). The discussions and relevance of GHG mitigation accounting methodologies and research needs presented in this chapter therefore have significance beyond just agroforestry and beyond just U.S. boundaries.

The GHG mitigation capacity of agroforestry will be influenced by how the trees, crops, livestock components, or a combination of the three are assembled into the many different agroforestry practices. The five main categories of agroforestry practices used in the United States are (1) riparian forest buffers, (2) windbreaks (including shelterbelts), (3) alley cropping (tree-based intercropping), (4) silvopasture, and (5) forest farming (multistory cropping), with a sixth category capturing adaptation of agroforestry technologies to address emerging issues such as biofeedstock production and stormwater management. A brief description of each agroforestry practice is provided in table 1.1 in chapter 1, along with a list of many potential benefits these practices may confer to the land and the landowner. Additional details on each of these practices and their potential benefits are available at the USDA National Agroforestry Center Web site (http://nac.unl.edu).

Approaches for assessing the GHG contributions at both entity (individual field or farm) and national scales are presented in this chapter. Relatively well defined and appropriate for forest and cropping/grazing systems, these approaches meet current Intergovernmental Panel on Climate Change (IPCC) *Guidelines for National Greenhouse Gas Inventories* (IPCC 2006) and provide a solid basis for constructing the consistent accounting methodologies across spatial scales needed in the more variable agroforestry systems. Although information is limited, agroforestry's other direct and indirect effects on GHG emissions (nitrous oxide [N_2O], methane [CH_4], and avoided emissions) are presented to inform future research and assessment activities required to build a more comprehensive understanding of agroforestry's contributions to agriculture's net GHG footprint and therefore how these practices can best be used within GHG mitigation strategies.

Potential Greenhouse Gas Mitigation Roles of Agroforestry

Temperate agroforestry is recognized as a viable land-management option for mitigating GHG emissions in the United States and Canada (CAST 2011, Eve et al. 2014, Nair et al. 2010, Schoeneberger et al. 2012). Agroforestry contributes to agricultural GHG mitigation activities by (1) sequestering carbon (C) in terrestrial biomass and soils, (2) reducing GHG emissions, and (3) avoiding emissions through reduced fossil fuel and energy usage. These GHG mitigation benefits are derived via the diversity of ecological functions created by and management activities used within agroforestry operations, both of which translate into greater C capture and tighter nutrient cycling in agroforestry compared with conventional operations under comparable conditions (Olson et al. 2000).

As a GHG mitigation tool, agroforestry can also provide additional ecosystem services and goods that producers and society value (see table 1.2 in chapter 1), including adaptive capacity for building added resiliency of operations and lands to changing climate (see chapter 2 of this assessment, ICF 2013, Nair et al. 2009, Plieninger 2011, Schoeneberger et al. 2012). As practiced in the temperate United States (with most practices, especially windbreaks and riparian forest buffers, comprising less than 5 percent of the field area), agroforestry can deliver these services while leaving the bulk of land in agricultural production. A large agricultural land base within the United States that could benefit from agroforestry includes nearly 22 percent of the cropland currently classified as marginal (ICF 2013). Even if only a small percentage of this area were converted, the potential C sequestration, along with other GHG reductions, could become noteworthy. When appropriately located, designed, and managed, agroforestry should not result in the conversion of additional lands into agricultural operations to replace these generally small portions of land now occupied in trees (see chapter 2 of this assessment, Schoeneberger 2009). For example, the 3 to 5 percent of a crop field put into a windbreak should result in increased yield (into the field up to a distance of 15 times the height of the trees), providing equal to greater returns from putting that small amount into trees (see the Commodity Production section in chapter 2).

Carbon Sequestration

Agroforestry's potential for sequestering large amounts of C is well recognized in both tropical and temperate regions (IPCC 2000, Kumar and Nair 2011, Plieninger 2011, Udawatta and Jose 2012). For example, 13-year-old poplar and spruce alley cropping systems in Canada had approximately 41 and 11 percent more total C, respectively, than accounted for in adjacent sole-cropping systems (Peichl et al. 2006). Positive trends in C sequestration have been documented in temperate regions, and the number of these studies is growing (see Kumar and Nair 2011). Tree-based agricultural practices tend to store more C in the woody biomass and soil compared with their treeless/more conventional agricultural alternatives under comparable conditions (table 3.1) (Lewandrowski et al. 2004, Nair 2012a). Similar to that observed in afforestation and reforestation activities (Gorte 2009), this C potential per unit area in agroforestry systems can be substantial, largely because of the amount of C sequestered in the woody biomass, with stem wood accounting for up to 90 percent of the new C (Hooker and Compton 2003).

Table 3.1. Estimated carbon sequestration rates for four main categories of U.S. agricultural land use.

Practice category	C sequestration rate (Mg of CO_2 eq ha^{-1} $year^{-1}$)
Afforestation (previously cropland or pasture)	6.7–19.0
Herbaceous riparian or conservation buffers	1.2–2.2
Conservation tillage (reduced to no-till)	0.7–1.7
Grazing management	2.7–11.9

CO_2 = carbon dioxide. eq = equivalent. ha = hectare. Mg = megagram.
Source: Data from Lewandrowski et al. (2004).

Based on U.S. land area that would be suitable for and benefit from the non-GHG mitigation services provided by agroforestry, such as soil- and water-quality protection (see chapter 2), estimates of agroforestry's C sequestration potential range from 90 teragrams (Tg) C per year (yr) (soil + biomass based at approximately 15 years into establishment) (Nair and Nair 2003) to 219 Tg C yr^{-1} (soil + biomass based at variable years [20 to 50 years] into establishment and depending on practice) (Udawatta and Jose 2012).

The range in estimates that have been reported over the years, from Dixon et al. (1994) to Udawatta and Jose (2012), vary substantially because of differences in (1) the assumptions used regarding C sequestering rates, (2) which pools were included in the estimate, (3) presumed project lifespans, and (4) the assumptions each study used to determine land area, where that land was, and how much of it would support each agroforestry practice type. The lack of national agroforestry inventory information in the United States limits our ability to estimate land area already under agroforestry and, therefore, current C sequestration contributions at regional and national scales (Perry et al. 2005, 2009; Robertson and Mason 2016). Regardless of the limited information, the data continue to affirm that we know enough to assess the direction of agroforestry's impact on C sequestration within an operation; that we know these impacts, in general, will be neutral to highly beneficial in comparison with more conventional operations; and that we know enough to estimate the larger C sinks in these systems (see CAST 2011, Kumar and Nair 2011, Schoeneberger 2009).

Because agroforestry is a combination of agricultural and forestry activities, the C stocks from which sequestered C is estimated should include the various pools from each of these activities. The size of these stocks (and sequestered C) will vary by agroforestry practice (fig. 3.1). These stocks will also vary with age and/or development of the woody component. Adding to the complexity of C fluxes within these system are the many interactions generated by the agroforestry plantings on other C components within the system, as illustrated in the windbreak example in figure 3.2. Recent work, such as by Wotherspoon et al. (2014), is helping to build a more comprehensive understanding of C fluxes generated by agroforestry—in this case, alley cropping; however, most studies to date report on only a portion of these pools. The lack of national inventory information (Perry et al. 2005) and the high cost and difficulty of collecting measurement information for all these pools have led to more pragmatic approaches for C research and accounting in agroforestry (Brown 2002, Schoeneberger 2009) (see discussion in table 3.2).

Carbon accounting within agriculture and forestry needs to consider five main pools: (1) live biomass (above ground), (2) live biomass (below ground), (3) dead biomass (dead

Figure 3.1. Continuum of agroforestry practices from agricultural field to forest stand, with relative carbon stocks by ecosystem pool associated with each practice. Note: This figure is for illustration purposes only; actual carbon stocks may vary widely, depending on the agroforestry prescription.

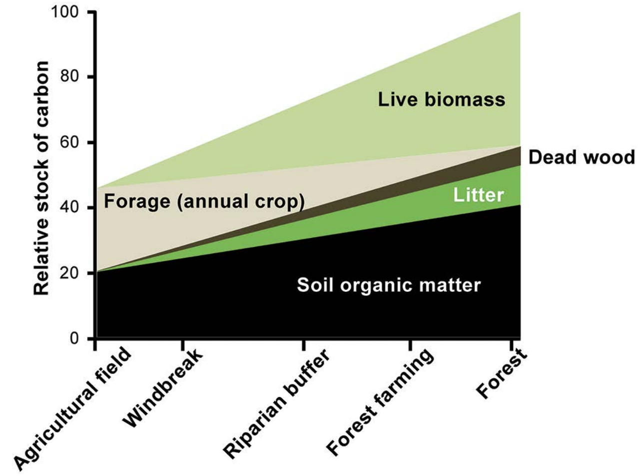
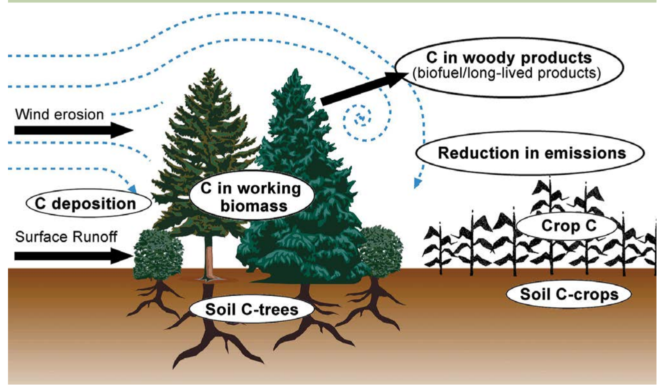


Figure 3.2. Major carbon sinks and sources that can be affected by a field windbreak. (Schoeneberger 2009).



wood), (4) dead biomass (forest floor), and (5) soil organic matter (Eve et al. 2014, IPCC 2006). Table 3.2 provides a brief overview regarding the feasibility of accurately and cost-effectively accounting for these various pools in agroforestry systems. Table 3.3 provides definitions for these pools, with references to estimation approaches for agricultural and forested systems. Depending on the level of specificity required for reporting, these pools may be further delineated, especially within forest land use (e.g., dead wood may be divided into standing and downed dead biomass; live biomass may be divided into live trees and understory biomass).

The majority of new aboveground C in agroforestry plantings is predominantly contained in the standing woody biomass, as documented in afforestation and reforestation plantings (Hooker and Compton 2003, Niu and Duiker 2006, Vesterdal et al. 2002). This component is the most readily visible, easily measured, and easily verified portion and, therefore, generally tends to be the more studied and reported component. Depending on the objectives, measures and/or estimates of belowground woody C and/or soil C may or may not be included with the agronomic component (Eagle and Olander 2012, Nair et al. 2010, Udawatta and Jose 2012).

Regarding the woody biomass component, Zhou et al. (2015) and Kort and Turnock (1999) demonstrated that use of existing allometric models obtained from forest-grown trees generally resulted in the underestimation of woody biomass and C in agroforestry plantings. Zhou et al. (2011, 2015) found that growth differences in specific gravity and architecture (both taper and ratio of stem to branch biomass) created in the more open canopies of agroforestry practices compared with forests contributed to this underestimation.

Less is known regarding C allocation to belowground woody biomass in agroforestry plantings. Estimation of this pool currently is accomplished pragmatically using forestry-derived protocols (Hoover et al. 2014). Ritson and Sochacki's work (2003) found more open-grown trees may have greater root biomass compared with close-spaced trees due to increased light and/or more root thickening in response to greater mechanical stress from wind sway. Taking these findings—for aboveground and belowground woody biomass and, therefore, C estimates—into account means that the generally neutral to very positive amounts of sequestered C reported to date for agroforestry practices in temperate regions may not fully reflect the whole contribution of this pool but can be considered a conservative assessment of agroforestry's C sequestration contribution.

Table 3.2. Accounting considerations for carbon sequestration pools in agroforestry plantings.[a]

Project effects	Ecosystem component	Contribution to reduction[b]	Flux[c]	Ability to measure or estimate[d]
Live biomass				
Above ground	Trees	++	L	**R—**Represents largest pool (Hooker and Compton 2003). Biomass equations should be modified for agroforestry plantings (Zhou et al. 2015) or regional biomass equations should be derived from forest stand data. The latter are currently available for most agroforestry species and will provide conservative (underestimated) values.
	Understory	+	S–M	**M—**Some plantings may potentially have a large shrub component, especially by design, so inclusion in accounting should be considered. Work is ongoing in the development of biomass estimation models for a few of the key species of shrubs used in agroforestry.
Below ground	Trees—coarse roots	++	M	**R—**Allometric equations should be used with root/shoot estimates (e.g., Birdsey 1992, Cairns et al. 1997). Increased partitioning of biomass/C to roots is observed in open-grown trees (Ritson and Sochacki 2003), so forest approaches will give conservative (underestimated) values for this component.
	Trees—fine roots	+	S–M	**N—**Turnover is extremely high, creating high variability and large error. Positive impact of this pool will be reflected, if at all, in the soil organic matter over time.
	Understory	+	S–M	**M—**Some plantings have the potential of having a large enough shrub component that inclusion in accounting should be considered. Work is ongoing in the development of biomass estimation models for a few of the key species of shrubs used in agroforestry.
Dead biomass				
Dead wood Forest floor		+	S	**N—**In an afforestation activity, like agroforestry, these components do not accumulate to significant levels until late in a practice's lifespan. Turnover of litter, in general, is higher in these more open systems. Variability and difficulty in estimating litter component is extremely high.
Soil organic matter				
Soil—carbon from biomass turnover		+	S	**M—**This matter can represent a pool that is influenced just under the tree component; variable distance from the tree over time (assuming main input from tree litter) or a combination of tree and crop management influences. Variability/error very high and with less certainty other than number will be positive in the long term. Soil accrual pool significantly smaller than that in woody biomass within row type plantings but could become quite significant in the plantings that occupy larger areas (i.e., alley cropping and silvopasture).

[a] Takes into account the woody portion of an agroforestry practice and the understory created by it. Does not include agricultural components altered by the integration of woody plants. Currently limited ability to account for the interactions between the agricultural and forestry components so are assessed separately. See discussion in the later part of this chapter.

[b] + and ++ = increasing positive net C potentially sequestered.

[c] S = small, M = medium, and L = large contribution to C sequestered in that pool, with S and M relative to the proportion the aboveground woody biomass comprises.

[d] R = recommended, M = maybe, and N = not recommended, based on lack of ease and reliability of getting the value and cost of measurement.

Source: Adapted from Schoeneberger (2009).

Table 3.3. Definitions of carbon pools that may exist in agroforestry practices and the data sources.

Carbon pool	Definition	Estimation approaches	
		Agricultural fields	Forest stands
Live biomass	**Live trees:** Large woody perennial plants (capable of reaching a height ≥ 15 feet) with a d.b.h. or at root collar (if multistemmed woodland species) ≥ 1 inch. Includes the C mass in roots (with diameters > 0.08 inches), stems, branches, and foliage.	Parton et al. (1987) Parton et al. (1998) Zhou (1999) Zhou et al. (2011) Zhou et al. (2015)	Smith et al. (2006) Jenkins et al. (2003) Woodall et al. (2011) Hoover et al. (2014)
	Understory: Roots, stems, branches, and foliage of tree seedlings, shrubs, herbs, forbs, and grasses.		Smith et al. (2006) Russell et al. (2014) Hoover et al. (2014)
Dead wood	**Standing dead:** Dead trees of ≥ 1 inch d.b.h. that have not yet fallen, including C mass of coarse roots, stems, and branches, but that do not lean more than 45 degrees from vertical, including coarse nonliving roots > 0.08 inches in diameter.		Smith et al. (2006) Harmon et al. (2011) Domke et al. (2011) Hoover et al. (2014)
	Downed dead: Nonliving woody biomass with a diameter ≥ 3 inches at transect intersection, lying on the ground. Also includes debris piles (usually from past harvesting) and previously standing dead trees that have lost enough height or volume or lean > 45 degrees from vertical, so they do not qualify as standing dead trees.		Smith et al. (2006) Hoover et al. (2014)
Litter	The litter layers and all fine woody debris with a diameter < 3 inches at transect intersection, lying on the ground above the mineral soil.		Smith et al. (2006) Hoover et al. (2014)
Soil organic matter	All organic material in soil to a depth, in general, of 3.3 feet, including the fine roots (< 0.08 inches in diameter) of the live and standing dead tree pools, but excluding the coarse roots of the pools above.	Del Grosso et al. (2001) Del Grosso et al. (2011) Ogle et al. (2003) Ogle et al. (2010) Parton et al. (1998)	Smith et al. (2006) Hoover et al. (2014)

d.b.h. = diameter at breast height. C = carbon.

Sources: Adapted from Eve et al. (2014); IPCC (2006).

Soil C stocks are likely to be altered in agroforestry plantings compared with conventional cropping or grazing systems in the United States, but the direction and magnitude of change will depend on the ecological context of the site and the type of agroforestry system implemented (see the Soil Resources section in chapter 2). Inherently highly variable, soil C has been found to be even more variable in agroforestry systems (e.g., Bambrick et al. 2010, Sharrow and Ismail 2004) compared with nearby forest-only plantation and treeless operations and may well explain the variability of agroforestry findings reported thus far. Methodological difficulties, including differences in sampling depth and selection of the site to provide comparative baselines, further limit discussion to qualitative rather than more quantitative comparisons, especially across regions, conditions, and different types of agroforestry practices (Nair 2012b). Results to date from agroforestry and afforestation studies in the United States indicate soil C sequestration under agroforestry may actually be negligible/undetectable to possibly negative for several years after initial establishment (Nave et al. 2013, Paul et al. 2003, Peichl et al. 2006, Udawatta et al. 2009).

Erosion control in agroforested areas also confounds easy and accurate assessments of C sequestration in the soil pool. Many agroforestry plantings, particularly windbreaks and riparian forest buffers, are purposely designed to intercept soil eroding from adjacent sources. These transported soils, either from wind erosion (Nuberg 1998, Sudmeyer and Scott 2002) or surface runoff (McCarty and Ritchie 2002), tend to be higher in C and other nutrients. The patterns of soil parameter data (i.e., litter mass, soil pH, and texture) measured by Sauer et al. (2007) from under a 35-year-old windbreak in Nebraska documented this deposition. Use of stable C isotope analysis is one means of separating out that C that is transported in and that C sequestered in situ. Hernandez-Ramirez et al. (2011), using this method, identified approximately 50 percent of the larger soil organic C (SOC) pool found beneath afforested areas versus adjacent cropland in Iowa was tree derived (1.73 kilograms [kg] C square meters [m^{-2}]), with an estimated mean residence time of 45 years and an estimated annual accrual rate of 10.6 grams C m^{-2} yr^{-1}. The cotransport of nitrogen (N) with eroded materials into the agroforestry planting may also cause confounding impacts. Although the addition of N via erosion has not been found to increase soil C efflux or deplete soil C stocks (Grandy et al. 2013, Janssens et al. 2010, Ramirez et al. 2012), it may be impacting soil C stocks in ways not yet identified or understood and requires further investigation.

Perhaps more substantial than the estimation of total SOC are the findings that these tree-based systems, compared with their treeless counterparts, tend to store significantly more C deeper in the profile and in the smaller sized fractions, all of which contribute a greater stability to this sequestered C (Haile et al. 2008, 2010; Howlett et al. 2011). Soils under the woody component of hedgerows, windbreaks, and silvopasture were found to consistently have greater total SOC and SOC in all size fractions when compared with the treeless agricultural component (Baah-Acheamfour et al. 2014).

GHG Mitigation of Other GHGs

Understanding agroforestry's broader role in GHG mitigation beyond the C sequestration described previously entails knowledge of its impacts on the other major GHGs of concern in agriculture, namely N_2O and CH_4. Research is limited regarding the impacts of agroforestry on these two GHGs. The tighter nutrient cycling created by the greater spatial and temporal diversity in agroforestry plantings (Olson et al. 2000) would support the premise that agroforestry should have neutral to beneficial effects in reducing emissions of these two GHGs when compared with conventional treeless practices under similar conditions. In addition, how other management activities, especially those involving the management of fertilizers and grazing, are implemented in the various agroforestry practices will also influence the direction and magnitude of this mitigation potential (box 3.1). The data and means for accurate estimates of these contributions are not available yet for building a quantitative understanding. Enough is now known, however, to identify the relative magnitude and direction of trends and also the mechanisms at play in the various agroforestry practices under different settings. Such information can assist in establishing improved design and management guidance that better optimize agroforestry's beneficial GHG functions.

GHG Mitigation of Nitrous Oxide

The potential to lower N_2O emissions in an alley cropping system (also referred to as tree-based intercropping system) was estimated at 1.2 kg N_2O hectare^{-1} yr^{-1} (Evers et al. 2010). Amadi et al. (2016) found N_2O emissions were about 4 times lower in shelterbelts, a factor they attributed to the exclusion of N fertilization and also possibly due to greater soil aeration under shelterbelt trees. Other studies have also documented N_2O emission reductions in tree plantings into crop and pasture lands; e.g., afforestation plantings (Allen et al. 2009), windbreaks (Ryszkowski and Kedziora 2007), and riparian forest buffers (Kim et al. 2009). Data regarding the magnitude of these trends are insufficient to judge the significance of these reductions at broader scales at this time (Ogle et al. 2014).

The elimination or reduction of N-fertilizer inputs on that portion of land planted to the agroforestry tree component will reduce N_2O emissions from that source. This amount can be estimated using methodologies described in Ogle et al. (2014). Tighter nutrient cycling that is generally observed in multistrata/multispecies plantings, such as agroforestry (Olson et al. 2000), should also play a role in reducing emissions, both on and off site. Observations of N conservation in agroforestry plantings as compared with treeless cropping and grazing systems have been documented (Allen et al. 2004, Bambo et al. 2009, López-Díaz et al. 2011, Nair et al. 2007). This effect will be altered depending on the various types and amounts of management activities implemented within the agroforestry system, as well as by the age of the woody plants (box 3.1).

N_2O emissions are influenced by many different factors and are highly variable in agricultural soils (Butterbach-Bahl et al. 2013, Eve et al. 2014). The additional complexity of spatial and temporal factors created in agroforestry, such as influence of tree development on nutrient cycling, is expected to play a role in reducing net N_2O emissions through greater soil N uptake; however, it will also make it more difficult to quantitatively assess. Riparian forest buffers warrant special attention regarding N_2O emissions. Although naturally occurring riparian forests have been identified as being potential N_2O hotspots—a function of intercepting additional N from runoff and having conditions conducive for denitrification (Groffman et al. 2000)—riparian forest buffers are those conservation plantings purposely designed and used to intercept field runoff, especially nitrates (NO_3), to protect water quality. Increased uptake of NO_3 by riparian forest buffer vegetation has the potential to reduce the amount of NO_3 that would otherwise be available for denitrification and subsequent N_2O emission (Kim et al. 2009, Tufekcioglu et al. 2003). Harvesting of plant materials in the riparian forest buffer zones closest to the fields would remove N from the site and help maintain more actively growing plant materials and therefore nutrient uptake, especially for N. To offset harvesting costs, these plant materials could be specifically selected for and then used/sold as a source of biofeedstock (Schoeneberger et al. 2008).

Box 3.1. Management activities within agroforestry practices that can potentially alter the magnitude and direction of carbon sequestration and other greenhouse gas fluxes.

Practice	Management activities

Windbreaks

- Disturbance to soil by site preparation during establishment.
- Deposition of wind- and water-transported sediments, nutrients, and other agricultural chemicals into the planting.
- Windbreak renovation (removal and replanting of dead and dying trees over time).

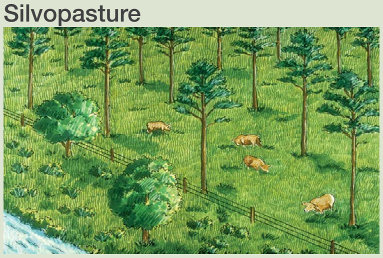
Riparian forest buffers

- Disturbance to soil by site preparation during establishment.
- Deposition of wind- and water-transported sediments, nutrients, and other agricultural chemicals into the planting.
- Harvesting of herbaceous materials planted in Zone 3 (zone closest to crop/grazing system) and of woody materials planted in Zone 2 (middle zone).

Alley cropping

- Disturbance to soil by site preparation during establishment.
- Weed control (mechanical or chemical).
- Pruning, thinning, and harvesting of woody material (amount and frequency vary greatly depending on short- and long-term objectives of practice).
- Fertilization for alley crop and possibly also occasionally for trees in rows (i.e., fruit/nut trees).
- Pesticides as needed for alley and row crops.
- Tillage in alleys (frequency and intensity).
- Crop species used in alley production.
- Complex harvesting schedules stratified in space and time.

Silvopasture



- Disturbance to soil by site preparation during establishment.
- Weed control (mechanical or chemical).
- Pruning, thinning, and harvesting of woody material (amount and frequency vary greatly depending on short- and long-term objectives of practice).
- Harvesting of needles for pine straw.
- Fertilization of forage component.
- Tillage in forage component (frequency and intensity).
- Crop species used in forage component.
- Grazing management (timing, intensity, frequency).
- Complex harvesting schedules stratified in space and time.

Forest farming

- Activities will be predominantly alterations of overstory for canopy manipulation and modification of understory as required for specific understory crop being grown and from harvesting of crops.

Special applications

- Special applications are essentially modifications of the above agroforestry practices to address issues such as urban stormwater treatment, biofeedstock production, and waste treatment and will entail similar activities as listed above but to varying levels and frequencies of applications.

Source: Adapted from Ogle et al. (2014).

GHG Mitigation of Methane

The second most prevalent GHG emitted in the United States from human activities, CH_4, is 25 times more efficient at trapping radiation than CO_2 (USDA-OCE 2016). The largest sources of CH_4 from agricultural activities are from livestock and manure management. The major strategies for reducing these emissions are therefore focused on altering how livestock and manure are managed; however, assessment of CH_4 flux in agroforestry is important to obtain a full GHG accounting of these systems within farm and ranch operations. Agroforestry can potentially alter the CH_4 emissions, albeit to only a small extent, by influencing microbially mediated soil activities responsible for CH_4 oxidation and reduction, with the latter most likely to occur at any measurable extent in well-aerated soils under upland practices and the former in periodically flooded soils that tend to occur in riparian environments. Agroforestry, specifically silvopasture, also can potentially influence CH_4 emissions but, in this case, through management and rotation of livestock.

Research findings for soil-mediated impacts in agroforestry systems at this stage are contradictory. Allen et al. (2009) and Priano et al. (2014) found CH_4 uptake to be greater in afforested ex-pasture sites than in pasture, suggesting that agroforestry that is afforestation-like could potentially influence CH_4 uptake positively. In a Canadian study, soil CH_4 oxidation potential under shelterbelts was 3.5 times greater than in cultivated soils, which was attributed to the trees creating more favorable moisture, soil organic matter, and infiltration conditions for CH_4 uptake (Amadi et al. 2016). Upland soils in general, particularly under forests, are identified as providing a CH_4 sink; however, this function generally is reduced by soil disturbance, such as tillage and N-fertilization (Dutaur and Verchot 2007, Suwanwaree and Robertson 2005, Topp and Pattey 1997). These findings would suggest that agroforestry practices, at least in the early years after establishment, may have only limited capacity for CH_4 oxidation activity. Soils in riparian forest buffers generally are found to be a CH_4 source due to anaerobic conditions created by periodic flooding. CH_4 flux in soils under established riparian vegetation in Iowa, however, did not differ from adjacent upland crop soils (Kim et al. 2010), most likely due to the altered hydrology generally encountered in these Midwest agricultural landscapes.

Silvopasture may have the greatest potential among the agroforestry practices to reduce CH_4 emissions. Livestock are the key CH_4 producers in silvopasture systems, and silvopasture affords several management opportunities to influence this production. Silvopasture introduces a grazing strategy of moving cattle in a rotational stocking system and has the potential to produce more digestible feed and greater overall gain from feed efficiency due to shade-induced microclimate changes (Cuartas et al. 2014, Lin et al. 1998, Mitlöhner et al. 2001) (see the Livestock Production section in chapter 2). Little GHG work has been done with all three silvopasture components in place (trees, forage, and livestock). Most studies have focused predominantly on only the C sequestering and nutrient uptake capacity of the tree and forage components in this system (e.g., Haile et al. 2010, Nair et al. 2007). Further work to integrate the animal component in the GHG modeling and accounting of silvopasture should be a priority, given the potential implication to reduce CH_4 emissions by improving forage quality via tree-based shading.

GHG Mitigation Through Emission Avoidance

Trees planted on agricultural lands and around farmsteads and facilities can increase feed efficiency, reduce the area of land tilled, and modify microclimate both around buildings—reducing heating/cooling needs—and near roads—reducing snow deposition and, therefore, snow removal on roads (Brandle et al. 1992, DeWalle and Heisler 1988, Kursten and Burschel 1993). These activities lead to reduced consumption of fossil fuels, chemical inputs that include N-fertilizer, and electricity and natural gas usage on farms and ranches, all of which lead to a reduction in GHG emissions. These reductions are also referred to as avoided emissions. Machinery fuel and oil, N, and herbicides, expressed in terms of kg of C equivalent (CE), have been estimated at 0.94 kg CE per kg fuel, 1.3 kg CE per kg of N-fertilizer, and 6.3 kg CE per kg of herbicide, respectively (Lal 2004). As proposed by Lal (2004), inclusion of energy use within the net GHG assessment of an operation provides a more complete picture for comparing farm and ranch management decisions.

Brandle et al. (1992) estimated potential C storage (sequestered carbon dioxide [CO_2]) and conservation (CO_2 avoided emissions) that might be realized in a United States-wide windbreak-planting program. Their findings indicate avoided emissions can play a greater role in GHG mitigation in agriculture than that realized from direct C sequestration via biomass. These estimates were based on broad assumptions and energy-efficiency conditions different from today. Further, they did not include a complete accounting of other potential contributions to avoided emissions (e.g., reduction in feed quantity required because of increased feed efficiency from livestock windbreaks). The magnitude of the estimated contributions found by Brandle et al. (1992), along with estimates from a more recent study (Possu 2015) strongly supports additional research in this area.

Emissions and Sequestration Accounting Methods

The IPCC (2006) *Guidelines for National Greenhouse Gas Inventories* presents two basic approaches—(1) the stock-difference method and (2) the gain-loss method—to emissions accounting and recommends using the method or combination of methods that provides the highest levels of certainty, while using the available resources as efficiently as possible. With the stock-difference method, mean annual net C emissions or sequestration for land subject to human activities is estimated as the ratio of the difference in C stock estimates at two points in time and the number of intervening years. With the gain-loss method, which is a process-based approach, annual changes in C stocks are estimated by summing the differences between the gains (e.g., increase in biomass) and losses (e.g., biomass decomposition) in a C pool. In the United States, both approaches are used to estimate C stock changes for different land uses, depending on the availability of inventory data. When inventory data exist (e.g., the national forest inventory from the USDA Forest Service Forest Inventory and Analysis [FIA] program), the stock-difference method is used. When inventory data are sparse, the gain-loss method or a combination of the two methods is used. In agroforestry systems, in which data are often limited, it is likely a combination of the two accounting methods will be used to obtain estimates of C stock changes in the woody and crop-related components. COMET-Farm (http://cometfarm.nrel.colostate.edu), a USDA Web-based tool for assessing GHG and C sequestration within farm and ranch operations, currently uses a stock-difference method in its Quick Agroforestry tool (for further discussion see box 6.1 in chapter 6 of this assessment).

Uncertainty

There is a need to develop agroforestry models with less uncertainty regarding C stocks and the other GHGs. The factors contributing to uncertainty in GHG accounting in agroforestry include measurement and sampling error, modeling error, and interpretation of the protocols one follows. Lack of data at both the entity and national scales is the primary source of uncertainty associated with estimates of GHGs in agroforestry systems (Nair 2012b). As new data become available, models specific to agroforestry systems may be developed that better reflect C stocks and stock changes in these environments.

Monte Carlo methods are often recommended for estimating the statistical uncertainty associated with GHG estimates (IPCC 2006). Although the methods may vary based on data availability, simulations generally are run many times (e.g., 1,000 to 10,000 times) to obtain a probability distribution around the GHG estimate of interest that can then be used to estimate statistical uncertainty. Part of the GHG research strategy for temperate agroforestry will need to take into account input requirements for such exercises.

Carbon Accounting at the Entity Level

One of the many potential benefits of agroforestry systems is the sequestration of CO_2 from the atmosphere in herbaceous and woody biomass and the accumulation of C in live and dead organic matter (IPCC 2000, Kumar and Nair 2011). Carbon accounting in agroforestry systems represents a challenge because of its mix of land use and management practices that intersect three distinct land-use categories: (1) forest land, (2) cropland, and (3) grassland (table 3.4) (EPA 2014). This section provides an overview of carbon pools and accounting approaches in agroforestry at the entity level (see Hoover et al. [2014] and Ogle et al. [2014] for a full description), with an emphasis on woody vegetation and associated ecosystem pools. The inventory and accounting methods described in this section are consistent with national and international protocols.

Table 3.4. Land-use categories used in GHG accounting in the United States that may include agroforestry practices.

Land-use category	Defining agency	Description
Forest land	USDA Forest Service (FIA program)	Land areas ≥ 36.6 m wide and 0.4 ha in size with ≥ 10 percent cover (or equivalent stocking) by live trees able to attain an in situ height of 5 m, including land that formerly had such tree cover and that will be naturally or artificially regenerated. Areas between forest and nonforest lands that have ≥ 10 percent cover (or equivalent stocking) with live trees and forest areas adjacent to urban and built-up land are also included. Areas such as shelterbelt strips of trees ≥ 36.6 m wide or 0.4 ha in size are also classified as forest.
Cropland	USDA Natural Resources Conservation Service (NRI program)	Land areas used for the production of agricultural crops for harvest, including both cultivated and noncultivated lands. Cultivated cropland includes row crops or close-grown crops and also hay or pasture in rotation with cultivated crops. Noncultivated cropland includes continuous hay, perennial crops (e.g., orchards), and horticultural crops. Cropland also includes land with alley cropping and windbreaks, and also lands in temporary fallow or enrolled in conservation reserve programs (i.e., set-asides), as long as these areas do not meet the forest land criteria.
Grassland	USDA Natural Resources Conservation Service (NRI program)	Land area composed principally of grasses, grass-like plants (i.e., sedges and rushes), forbs, or shrubs suitable for grazing and browsing and includes both pastures and native rangelands. Includes areas where practices such as clearing, burning, chaining, and/or chemicals are applied to maintain the grass vegetation; savannas, some wetlands, deserts, and tundra; woody plant communities of low forbs and shrubs, such as mesquite, chaparral, mountain shrub, and pinyon-juniper, if they do not meet the criteria for forest land; and land managed with agroforestry practices such as silvopasture and windbreaks, assuming the stand or woodlot does not meet the criteria for forest land.

FIA = Forest Inventory and Analysis. GHG = greenhouse gas. ha = hectare. m = meter. NRI = Natural Resources Inventory. USDA = U.S. Department of Agriculture.

System Boundaries and Scale

The inventory and accounting methods described in this section have been modified from strategic level guidelines (IPCC 2006, Smith et al. 2006) for use at the entity level.

C fluxes will occur across the system boundary; however, they are not typically estimated, with the exception of harvested wood products. Given the array of agroforestry practices, trees on a property may not fit a particular land-use definition, creating complexities in inventory and accounting. Methods from multiple land-use categories (i.e., forest land, cropland, and grassland) will likely be required—with care taken to avoid double counting—to obtain a comprehensive estimate of C stocks and stock changes for the entity (see Eve et al. [2014] for complete descriptions of accounting techniques for different land-use categories).

Unlike annual crops, which are considered by the IPCC (2006) and the U.S. Environmental Protection Agency (EPA 2014) to be ephemeral with no net emission to the atmosphere (West et al. 2011), perennial woody crops have the potential to sequester large amounts of C per unit area (Dixon et al. 1994, Kumar and Nair 2011, Nair et al. 2010). To account for C stocks and stock changes in agroforestry systems, measurements collected as part of a field inventory may be used to meet the necessary data requirements for C accounting purposes. In most cases, repeated annual measurements are not practical, nor are the changes in C stocks sufficiently different from year to year to support such remeasurements. Instead, models and/or lookup tables from the IPCC (2006) and Eve et al. (2014) may be needed to account for temporal changes in vegetation and associated ecosystem pools when longitudinal datasets are not available.

Summary of Inventory and Data Requirements

Inventories of natural resources contribute to the accounting of various products and/or services (e.g., C sequestration) those resources provide. In agroforestry systems, C pools may be broadly or narrowly defined, depending on the size of the entity and type of management practice. Systems (e.g., forest farming) that resemble forest stands may include all ecosystem pools typically associated with forest land, but practices in which trees are a minor component (e.g., alley cropping and windbreaks) may include only certain ecosystem pools common in forest stands (fig. 3.1). The type of agroforestry system will dictate which accounting methods are used to obtain C stock and stock-change estimates and the inventory information necessary to compile those estimates (Eve et al. 2014).

Estimation of C Pools

Obtaining sound estimates of C stocks and stock changes in agroforestry systems requires balancing data availability with the entity's resources and needs. Explicitly establishing system boundaries and the C in ecosystem pools to be included in the accounting framework will help identify possible gaps or overlaps between pools or methods, particularly when combining methods across land-use categories. Furthermore, consistent definitions and estimation methods must be used for each pool to ensure valid estimates of C stock changes (Eve et al. 2014, IPCC 2006).

Many of the estimation and sampling approaches used to account for C in agroforestry systems come from the forestry and agricultural literature. As such, it may be helpful when identifying estimation and sampling strategies to think about agroforestry practices and, even just within an agroforestry practice, as occurring within an agriculture and forestry continuum (figs. 3.1 and 3.2). Carbon in agroforestry practices that are dominated by agricultural crops (e.g., windbreaks) may be best accounted for using approaches developed for agricultural applications. Carbon in agroforestry practices that more closely resemble forest conditions (e.g., forest farming) may be best accounted for using methods developed in forestry. In other words, the distribution of C from agricultural fields to forest will dictate which models, measurements, and sampling design one chooses to quantify C stocks and stock changes.

This section focuses on C in perennial crops and soil organic matter as annual crops (i.e., most food crops and some forages, such as rye, oats, and wheat) are not typically included in C accounting. Live perennial biomass therefore includes live trees (above and below ground), shrubs, seedlings, and herbaceous vegetation (table 3.4). Some or all of these components of the live biomass pool may exist in agroforestry practices and in forest conditions; this pool accounts for as much as one-half of the C storage (EPA 2014). Dead wood includes standing dead trees and downed dead wood (table 3.4). Dead wood may be a negligible component of many agroforestry practices, but, in systems managed to more closely resemble forest conditions, one or both of these components may exist and be important contributors to the C stocks and fluxes. Litter and fine woody debris (table 3.4) are small but important components in forests and, although they may be minor components in agroforestry systems, approaches for estimating this ecosystem pool exist. Finally, SOC (table 3.4) is a major component in forests and agricultural landscapes and accounts for a substantial amount of C storage in these systems (EPA 2014).

Although inventory and sampling methodologies are beyond the scope of this chapter (see Pearson et al. [2007] for a description of C inventories), each ecosystem pool mentioned may exist in an entity-level accounting framework in agroforestry systems. For a complete description of entity-level accounting in agroforestry systems as it currently stands, see Eve et al. (2014).

Carbon Accounting at the Regional and National Levels

This section provides an overview of the assessment of C emissions and sinks resulting from the uses and changes in land types and forests in the United States, with emphasis on agroforestry systems. The IPCC *Guidelines for National Greenhouse Gas Inventories* (IPCC 2006) recommends reporting fluxes according to changes within and conversions between certain land-use types termed forest land, cropland, grassland, and settlements (and also wetlands). Agroforestry practices under the current organization of land-use categories in the U.S. National Greenhouse Gas Inventory (NGHGI) report are not explicitly characterized and, given the complex of land uses, may not be included in the national inventories used to compile C stocks and stock-change estimates for the United States (Perry et al. 2005, 2009). That said, agroforestry systems may be represented within the forest land (e.g., shelterbelts), cropland (e.g., alley cropping, windbreaks), or grassland (e.g., windbreaks, silvopasture) land-use types in the NGHGI if they meet the minimum definitions for each land use defined by the national inventories.

Land Representation in National Accounting

In accordance with IPCC (2006) guidelines for reporting GHG fluxes to the United Nations Framework Convention on Climate Change, the United States uses a combination of approaches and data sources to (1) determine areas of managed and unmanaged lands, (2) apply consistent definitions for the land-use categories over space and time, and (3) account for all C stock changes and non-CO_2 GHG emissions on all managed lands (EPA 2014). Aspatial data from the Natural Resources Inventory (NRI) and FIA programs are used with spatially explicit time series land-use data from the National Land Cover Database (NLCD) to provide a complete representation of land uses and land-use change for managed lands. In general, land in the United States is considered managed if direct human intervention has influenced its condition and all other land is considered unmanaged (EPA 2014).

IPCC (2006) identifies six main land-use categories. In the United States, land-use definitions are country specific and are consistent with those used in the NRI and FIA programs. Agroforestry systems represent a complex of land use and management practices that intersect three distinct land-use categories in the NGHGI: (1) forest land, (2) cropland, and (3) grassland (table 3.4).

National Accounting Data Sources

The different land uses are monitored by national inventory programs that focus primarily on forest lands and agricultural lands. Because certain agroforestry practices may not meet the definitions of the different land uses used in national inventory programs, they may not be monitored (Perry et al. 2005). As a result, there is not sufficient data to characterize C stocks and stock changes at a national scale for certain agroforestry practices as required in national and international C reporting instruments. That said, the FIA program has several pilot studies currently under way to evaluate novel approaches to monitoring remote areas (e.g., interior Alaska), urban ecosystems, and tree cover in agricultural landscapes (Liknes et al. 2010, Meneguzzo et al. 2013).

Natural Resources Inventory

The NRI is the official source of data on all land uses on non-Federal lands in the conterminous United States and Hawaii (except forest land), and it is also used as the resource to determine the total land base for the conterminous United States and Hawaii. The NRI is a statistically based survey conducted by the USDA Natural Resources Conservation Service and is designed to assess soil, water, and related environmental resources on non-Federal lands. The NRI survey uses data obtained from remote-sensing imagery and field visits to provide detailed information on land use and management, particularly for croplands and grasslands, and is used as the basis to account for C stock changes in agricultural lands (except Federal grasslands).

Forest Inventory and Analysis

The FIA program, conducted by the USDA Forest Service, is another statistically based survey for the United States; it is the official source of data on forest land area and management. The FIA program employs a three-phase annual inventory, with each phase contributing to the subsequent phase. Phase 1 is a variance-reduction step in which satellite imagery is used to assign Phase 2 (P2) plots to strata (Bechtold and Patterson 2005). P2 plots are distributed approximately every 2,428 ha across the 48 conterminous States of the United States and are visited every 5 to 10 years (i.e., 10 to 20 percent of plots are remeasured in each State each year). Each P2 permanent ground plot comprises a series of smaller fixed-radius plots (i.e., subplots) spaced 36.6 m apart in a triangular arrangement, with one subplot in the center. Tree- and site-level attributes—such as diameter at breast height and tree height—are measured at regular temporal intervals on P2 plots that have at least one forested condition (USDA Forest Service 2013). Every 16th P2 plot is a Phase 3 plot where additional attributes on live and dead trees, forest floor, understory vegetation, and soils are sampled. This information is used to estimate C stocks and stock changes on managed forest land (i.e., direct human intervention has influenced its condition) in the United States.

National Land Cover Database

The NLCD is used as a supplementary database to account for land use on Federal lands (e.g., Federal grasslands) that are not included in the NRI and FIA databases. The NLCD land-cover classification scheme, available for 1992, 2001, 2006, and 2011, has been applied over the conterminous United States (Homer et al. 2004) and also for Alaska and Hawaii in 2001. For the conterminous United States, the NLCD Land Cover Change Products for 2001 and 2006 were used to represent both land use and land-use change for Federal lands (Fry et al. 2011, Homer et al. 2004). The NLCD products are based primarily on Landsat Thematic Mapper imagery. The NLCD is strictly a source of land-cover information and does not provide the necessary site conditions, crop types, and management information from which to estimate C stock changes or GHG emissions on those lands.

Carbon Stocks and Stock Changes

The relevant land-use categories that may include agroforestry systems or the C pools that comprise agroecosystems include forest land, croplands, and grasslands. This section provides an overview of the estimation methods for C stocks and stock changes within the C pools relevant in agroforests by land-use category.

Forest Land

Five C pools are defined by the IPCC (2006) for estimating C stocks or stock changes in forest ecosystems. These pools are consistent with the pools defined in table 3.3, although live biomass is separated into aboveground and belowground components for national reporting. Forest ecosystem stock and flux estimates are based on the stock-change method, and calculations for all estimates are in units of C. Separate estimates are made for the five storage pools. All estimates are based on data collected from FIA plots and from models employed to fill gaps in field data (USDA Forest Service 2013). Carbon-conversion factors are applied at the disaggregated level of each inventory plot and then appropriately expanded to population estimates. A combination of tiers as outlined by IPCC (2006) is used. The Tier 3 biomass C values are calculated from FIA tree-level data. The Tier 2 dead organic and soil C pools are based on land use, land-use change, and forestry empirical or process models from FIA data. All C-conversion factors are specific to regions or individual States within the United States, which were further classified according to characteristic forest types within each region.

Croplands

Changes in soil C stocks due to agricultural land use and management activities on mineral soils and organic soils are estimated according to land-use histories recorded in the USDA NRI survey (USDA-NRCS 2009). An IPCC Tier 3 model-based approach (Ogle et al. 2010) was applied to estimate C stock changes for mineral soils used to produce most annual crops (e.g., alfalfa hay, barley, corn, cotton, dry beans, grass hay, grass-clover hay, oats, onions, peanuts, potatoes, rice, sorghum, soybeans, sugar beets, sunflowers, tomatoes, and wheat) in the United States in terms of land area. The model-based approach uses the DAYCENT biogeochemical model (Del Grosso et al. 2001, 2011; Parton et al. 1998) to estimate soil C stock changes and soil N_2O emissions from agricultural soil management. Coupling the two source categories in a single inventory analysis ensures a consistent treatment of the processes and interactions between C and N cycling in soils. The remaining crops on mineral soils were estimated using an IPCC Tier 2 method (Ogle et al. 2003). The Tier 2 method was also used for very gravelly, cobbly, or shaley soils (greater than 35 percent by volume). Mineral SOC stocks were estimated using a Tier 2 method for these areas because the DAYCENT model, which is used for the Tier 3 method, has not been fully tested for estimating C stock changes in certain cropping systems. An additional stock-change calculation was estimated for mineral soils using Tier 2 emission factors to account for enrollment patterns in the USDA Conservation Reserve Program after 2007, which was not addressed by the Tier 3 method.

Annual C emissions from drained organic soils in cropland are estimated using the Tier 2 method provided in IPCC (2006), with U.S.-specific C loss rates (Ogle et al. 2003) rather than default IPCC rates.

Grasslands

Changes in soil C stocks due to agricultural land use and management activities on mineral and organic soils for private grasslands are estimated according to land-use histories recorded in the USDA NRI survey (USDA-NRCS 2009). Land use and some management information (e.g., crop type, soil attributes, irrigation) were originally collected for each NRI point on a 5-year cycle beginning in 1982. In 1998, the NRI program initiated annual data collection, and the annual data are currently available through 2010. NRI points were classified as “grassland remaining grassland” back to 1990 (the baseline year) if the land use had been grassland for 20 years. Grassland includes pasture and rangeland used for grass forage production, where the primary use is livestock grazing. Rangelands are typically extensive areas of native grassland that are not intensively managed, while pastures are often seeded grassland, possibly following tree removal, that may or may not be improved with practices such as irrigation and interseeding legumes.

An IPCC Tier 3 model-based approach (Ogle et al. 2010) is applied to estimate C stock changes for most mineral soils in non-Federal grasslands remaining grasslands. The C stock changes for the remaining soils are estimated with an IPCC Tier 2 method (Ogle et al. 2003), including gravelly, cobbly, or shaley soils (greater than 35 percent by volume) and additional stock changes associated with sewage sludge amendments. Annual C emissions from drained organic soils in grasslands are estimated using the Tier 2 method provided in IPCC (2006), which uses U.S.-specific C loss rates (Ogle et al. 2003) rather than default IPCC rates, as described in the Cropland Remaining Cropland section for organic soils (IPCC 2006).

Advancing Greenhouse Gas Performance and Accounting in Agroforestry Systems

Agroforestry systems are purposely diverse and complex, deliberately mixing both forestry and agriculture components into a variety of practices, a variety of designs (e.g., species compositions, arrangements), and a variety of settings, and involving a variety of other forestry and agricultural management activities (i.e., fertilization, harvesting, grazing, and tillage) (box 3.1). The ability of agroforestry to confer its many ecosystem benefits from production to landscape health is attributable to this high functional and structural diversity (Olson et al. 2000). All these factors influence how much C can be sequestered and GHGs emitted or avoided, which means agroforestry affords us a very flexible and potentially powerful arrangement of options to improve GHG mitigation performance, along with other functions being sought from agriculture by producers and society.

Capitalizing on Agroforestry's Complexity

Understanding how best to utilize these plantings in GHG mitigation strategies requires that we can understand, document, and account for all of agroforestry's GHG impacts. Agroforestry practices are expressly designed to capitalize on the beneficial interactions generated among the tree, crop, and livestock components, the impacts of which may occur well beyond the area occupied by the agroforestry planting itself. For example, a crop windbreak is designed to favorably modify microclimate on the adjacent crop field, an impact that can extend up to a distance of 15 times the windbreak tree height (Brandle et al. 2009). Windbreak-induced shifts in crop growth and soil microclimate in the field adjacent to the practice can then potentially further alter soil C fluxes and N_2O emissions in the field. We must actually look beyond the C sequestered in the wood and soil just under the trees, as is done now, to fully capture agroforestry's C benefits (fig. 3.2).

Interpractice soil C transfers also need to be considered in agroforestry GHG accounting. Many agroforestry plantings are explicitly designed to intercept or alter wind- and water-borne soil erosion, a climate change adaptive function that is predicted to become more critical under future weather events (see the Soil Resources section in chapter 2). Higher levels of soil C under windbreaks in the Great Plains and elsewhere are partially attributable to this interception of wind-blown soils (Sauer et al. 2007). These windbreak-intercepted soils have also been generally found to be richer in C (Sudmeyer and Scott 2002). Increased soil movement from upland fields into a riparian wetland area was associated with increased C sequestration rates in a riparian wetland (McCarty and Ritchie 2002). The more limited management and therefore more limited disturbance within riparian areas also suggest riparian forest buffers can serve as a longer term sink for C in this landscape. These erosional processes also deliver N into agroforestry practices, which is expected to influence N_2O flux in many different ways, depending on landscape position, site conditions, and vegetation and to also then impact C dynamics.

Species selections and planting configurations and densities are key considerations in designing for GHG mitigation-enhanced services from agroforestry. For instance, use of fast-growing species such as hybrid poplar can provide rapid C sequestration and N uptake, albeit with a shorter project duration than using slower growing species. Slower growing species, on the other hand, may be selected for the very purpose of longer function, such as in a windbreak, and thus have a longer project duration in which to sequester C. Mixtures of species, such as the herbaceous and woody plants used in riparian forest buffers, may be selected to optimize these GHG factors on site and also to provide other opportunities like biofeedstock production, which, in turn, would have additional GHG benefits (fig. 9.1 in chapter 9 of this assessment) (Schoeneberger et al. 2008). Other considerations can involve timing, placement, and type of N-fertilizer in agroforestry practices, where needed, and animal stocking numbers and rotation lengths in silvopasture systems (table 3.3). Many considerations can go into the planning and design of agroforestry, GHG mitigation being potentially one of them.

The various roles agroforestry can play in both GHG mitigation and climate change adaptation in U.S. agriculture—all depending on design and management—affords us the opportunity to rethink these practices in terms of optimizing benefits across the multiple objectives being sought for these lands (Schoeneberger et al. 2012). Waterbreaks are one example. A waterbreak is a planned floodplain system of linear woody buffers oriented to reduce flooding impacts (Wallace et al. 2000). Properly designed and located, waterbreaks could help address the potential impacts of the increased frequency and intensity of flood events being predicted under climate change and can also provide enhanced GHG mitigation services and many other nonflood-related services (fig. 9.3 in chapter 9 of this assessment). Other examples are presented in chapter 9.

Accounting Needs

Advancements are being made in the approaches for determining relative values and directions of GHG impacts from agroforestry (see Ogle et al. 2014). Tools, like USDA COMET-Farm (http://www.cometfarm.nrel.colostate.edu), now available for entity-level C reporting and planning, have incorporated modules for agroforestry and other woody plantings. These tools can help land managers compare the relative amounts of GHG mitigation from the many different climate-smart management options available, including agroforestry.

As mentioned previously, although current GHG accounting in agroforestry is focused on C in the woody biomass and in the soil under the woody plants, full GHG accounting will need to take into account that (1) the agroforestry-influenced unit may be greater than just the agroforestry-planted area (fig. 3.2), and (2) spatial and temporal factors will need to be considered for the mixture of components (fig. 3.1 and fig. 3.3). For example, silvopasture requires not only the modification and coupling of crop and forestry accounting approaches, but also inclusion of accounting for the livestock. Again, at this time, accounting can essentially only be done for each individual component as if no interactions occur among the components. Work by Dube et al. (2011) in silvopasture, which involves the co-management of trees, forage, and animals, provides us a glimpse of the many integrated GHG dynamics in these highly integrated systems.

As an agricultural management activity, agroforestry GHG information needs are similar to those already identified for agriculture in general (Olander et al. 2013) and include—

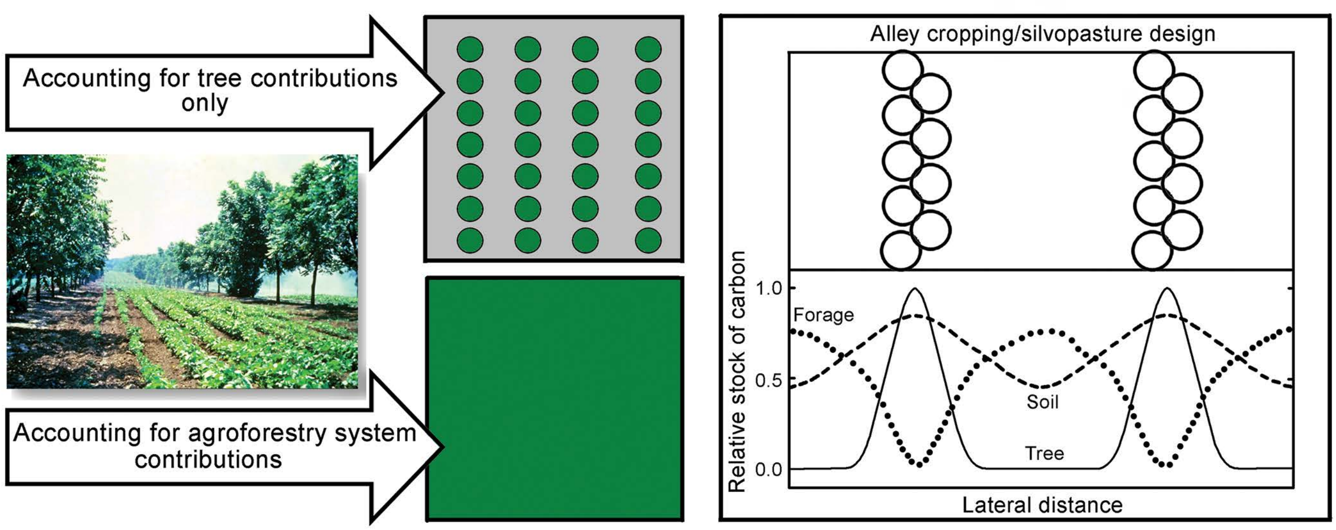
- User-friendly methods that work across scales, regions, and systems.
- Lower cost, feasible (end users' willingness to use) approaches.
- Methods that can crosswalk between emission-reduction strategies and inventories for reporting.
- Easily understood and common metrics for policy and market users.
- Continued research to account for and address the uncertainties in all the previous needs.

Although agroforestry practices have been part of the landscape for hundreds of years, they now reflect a wide variety of forms, management activities, and geographic settings. Performing the number of studies needed to adequately describe the performance of an agroforestry practice is physically and economically difficult. Regional and national coordination of agroforestry studies provides a more effective means to generate the necessary data. Standardization of measurement and modeling protocols would allow studies to be directly compared and the data then to be aggregated for additional research analyses and modeling efforts.

A Common Framework for Greenhouse Gas Accounting in Agroforestry

GHG assessments of agriculture's many activities, including agroforestry, need to be compatible for maximum use of the data collected (i.e., to compare between activities and to aggregate the contributions of many activities into a whole-farm context) (Olander et al. 2013). To this end, the report *Quantifying Greenhouse Gas Fluxes in Agriculture and Forestry: Methods for Entity-Scale Inventory* was developed to create an

Figure 3.3. Complexities of carbon sequestration accounting within an agroforestry practice as illustrated by an alley cropping/silvopasture design. Accounting must be pragmatic, however, with the acknowledgment that accounting in agroforestry is not a 1+1=2 system but rather one in which 1+1 may be either greater than or less than 2, depending on the spatial and temporal factors influencing these interactive and long-lived systems.



updated standard set of GHG estimation methods for use by USDA, landowners, and other stakeholders to assist in GHG management decisions (Eve et al. 2014). Key considerations are consistent with IPCC (2006) and include—

- ***Transparency.*** Clearly explained assumptions and methodologies to facilitate replication.
- ***Consistency.*** Methods and estimates internally consistent between years and, to the extent possible, with other USDA inventory efforts.
- ***Comparability.*** Estimates of emissions and sequestration reported by one entity should be comparable to those reported by others.
- ***Completeness.*** Must account for all sources and sinks and also for all GHG to the greatest extent possible.
- ***Accuracy.*** Accurate estimates that are systematically neither over nor under true emissions or removals as far as can be judged.
- ***Cost-effectiveness.*** Balance between the relative costs and benefits of additional efforts to improve the inventory or reduce uncertainty.
- ***Ease of Use.*** Level of complexity of the user interface and underlying data requirements.

These considerations are especially relevant to agroforestry efforts in the United States. Efforts to build regional understanding and GHG accounting of agroforestry in the United States are currently limited by not only a lack of data but also by disparate sampling protocols and designs used between studies (Nair 2012b). A more coordinated approach that could be used among the agroforestry researchers within the United States, other North American countries, and other temperate regions would create a more cost-effective strategy for generating the data needed to inform GHG and climate change decision-making (Nair 2012b, Schoeneberger et al. 2012).

A logical place to begin framing a common approach to GHG assessment in agroforestry is perhaps best placed in the land use into which it is primarily deployed—agriculture. Such a coordinated approach is the USDA Agricultural Research Service's GRACEnet (Greenhouse gas Reduction through Agricultural Carbon Enhancement network) effort (Liebig et al. 2012). GRACEnet provides a national framework for standardized approaches to assess C sequestration and GHG emissions from different cropping and rangeland systems, using common measurement protocols and coordinated regional experimental design (Walthall et al. 2012). By capitalizing on GRACEnet's already-established framework and protocols, data generated should be readily comparable across agroforestry studies and practices as well as across the many other agricultural management practices, thereby enabling a more accurate whole-farm accounting.

Capitalizing on Agroforestry's GHG and Adaptation Benefits

Pursuing agroforestry-derived GHG mitigation and climate change adaptation services simultaneously has technical and financial advantages (Duguma et al. 2014, Motocha et al. 2012, Plieninger 2011). GHG mitigation by agroforestry is dependent on having the plantings in place; however, adoption of agroforestry will be dependent on its cost-effectiveness, for whatever reason. Capitalizing on both the mitigation and adaptation services agroforestry can provide may help tip the balance in terms of cost-effectiveness for establishing new plantings. Carbon payments alone may influence adoption of agroforestry. However, the additional incentives tied to attainment of the adaptive services and goods agroforestry can also provide, such as protecting soil and air quality and providing critical wildlife habitat (e.g. pollinators), could lower the break-even prices even further and lead to greater adoption by farmers and ranchers (ICF 2013). Agroforestry also has the potential to generate additional income through diversified production and through hunting and other recreational fees, providing additional incentive. Use of these plantings as GHG mitigation strategies will ultimately hinge on the economics of agroforestry use (see chapter 4 in this assessment for further discussion of financial considerations regarding agroforestry) and on other producer values (see chapter 5 in this assessment for further discussion regarding adoption of agroforestry).

Key Findings

- Agroforestry plantings can sequester C in soils and biomass and mitigate other GHG emissions while leaving the bulk of land in agricultural production and providing other production, natural resource, and climate change adaptation services.
- The C sequestration and indirect C (emission avoidance) benefits from agroforestry systems are generally comparable or larger in magnitude than many other agricultural management activities. With high rates of C sequestration per unit area, even small plantings like windbreaks can provide substantial contributions to whole-farm GHG mitigation.
- Agroforestry's other GHG mitigation services, while not all fully understood, appear to also contribute to the improvement of the GHG footprint of individual farm and ranch operations.
- The specifics of agroforestry design and management activities influence the amounts and duration of C sequestration and potential reduction in GHG emissions. As such, agroforestry, with its many components, provides a highly flexible and versatile management option to improve GHG mitigation and production services.

Key Information Needs

- Identification of land in the United States suitable, both biophysically and cost-effectively, for establishing the various agroforestry practices to optimize GHG benefits along with other services agroforestry can provide.
- A national inventory to cost-effectively track land currently in agroforestry with a description of plantings (e.g., practice, age, condition) over time to evaluate contributions and include within U.S. GHG inventory assessments.
- A common GHG assessment framework to efficiently advance measurement, understanding, and predictive capacity of agroforestry's GHG services across the range of spatial and temporal settings in which agroforestry can be placed in the United States.
- Refined tools and methodologies for cost-effective and verifiable measurements/estimations of agroforestry's long-term potential to mitigate GHG emissions within the many agricultural production systems across the United States.
- Criteria and design tools to assist producers in developing appropriate configurations, species selections, and planting densities in the various agroforestry practices that optimize GHG mitigation along with other ecosystem services, including adaptation of and by the plantings to extreme weather events and other climate change impacts.

Literature Cited

Allen, D.E.; Mendham, D.S.; Singh, B. [et al.]. 2009. Nitrous oxide and methane emissions from soil are reduced following afforestation of pasturelands in three contrasting climatic zones. Australian Journal of Soil Research. 47(5): 443–458.

Allen, S.; Jose, S.; Nair, P.K.R. [et al.]. 2004. Safety net role of tree roots: experimental evidence from an alley cropping system. Forest Ecology and Management. 192(2): 395–407.

Amadi, C.C.; Van Rees, K.C.J.; Farrell, R.E. 2016. Soil-atmosphere exchange in shelterbelts compared with adjacent cropped fields. Agriculture, Ecosystems and Environment. 223: 123–124.

Baah-Acheamfour, M.; Carlyle, C.N.; Bork, E.W.; Chang, S.X. 2014. Trees increase soil carbon and its stability in three agroforestry systems in central Alberta, Canada. Forest Ecology and Management. 328: 131–139.

Bambo, S.K.; Nowak, J.; Blount, A.R. [et al.]. 2009. Soil nitrate leaching in silvopastures compared with open pasture and pin plantation. Journal of Environmental Quality. 38(5): 1870–1877.

Bambrick, A.D.; Whalen, J.K.; Bradley, R.L. [et al.]. 2010. Spatial heterogeneity of soil organic carbon in tree-based intercropping systems in Quebec and Ontario, Canada. Agroforestry Systems. 79(3): 343–353.

Bechtold, W.A.; Patterson, P.J. 2005. The enhanced Forest Inventory and Analysis program: national sampling design and estimation procedures. Gen. Tech. Rep. SRS-GTR-80. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 85 p.

Brandle, J.R.; Hodges, L.; Tyndall, J.; Sudmeyer, R.A. 2009. Windbreak practices. In: Garrett, H.E., ed. North American agroforestry: an integrated science and practice. 2nd ed. Madison, WI: American Society of Agronomy: 75–104. Chap. 5.

Brandle, J.R.; Wardle, T.D.; Bratton, G.F. 1992. Opportunities to increase tree plantings in shelterbelts and the potential impacts on carbon storage and conservation. In: Sampson, R.N.; Hairs, D., eds. Forests and global change. Vol. 1. Washington, DC: American Forests: 157–175. Chap. 9.

Brown, S. 2002. Measuring, monitoring, and verification of carbon benefits for forest-based projects. Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences. 360(1797): 1669–1683.

Butterbach-Bahl, K.; Baggs, E.M.; Dannenmann, M. [et al.]. 2013. Nitrous oxide emissions from soils: How well do we understand the processes and their controls? Philosophical Transactions of the Royal Society B: Biological Sciences 368(1621): 1–13. http://rstb.royalsocietypublishing.org/content/royptb/368/1621/20130122.full.pdf. (20 May 2016).

Cairns, M.A.; Brown, S.; Helmer, E.H.; Baumgardner, G.A. 1997. Root biomass allocation in the world's upland forests. Oecologia. 111(1): 1–11.

Council for Agricultural Science and Technology (CAST). 2011. Carbon sequestration and greenhouse gas fluxes in agriculture: challenges and opportunities. CAST Task Force Report 142. Ames, IA: Council for Agricultural Science and Technology. 106 p.

Cuartas, C.A.; Naranjo, J.F.; Tarazona, A.M. [et al.]. 2014. Contribution of intensive silvopastoral systems to animal performance and to adaptation and mitigation of climate change. Revista Colombiana de Ciencias Pecuarias. 27(2): 76–94.

Del Grosso, S.J.; Parton, W.J.; Keough, C.A.; Reyes-Fox, M. 2011. Special features of the DayCent modeling package and additional procedures for parameterization, calibration, validation, and applications. In: Ahuja, L.; Ma, L., eds. Methods of introducing system models into agricultural research. Madison, WI: American Society of Agronomy; Crop Science Society of America; Soil Science Society of America: 155–176.

Del Grosso, S.J.; Parton, W.J.; Mosier, A.R. [et al.]. 2001. Simulated interaction of carbon dynamics and nitrogen trace gas fluxes using the DAYCENT model. In: Schaffer, M.; Ma, L.; Hansen, S., eds. Modeling carbon and nitrogen dynamics for soil management. Boca Raton, FL: CRC Press: 303–332.

Denef, K.; Archibeque, S.; Paustian, K. 2011. Greenhouse gas emissions from U.S. agriculture and forestry: a review of emission sources, controlling factors, and mitigation potential. Interim report to the U.S. Department of Agriculture. http://www.usda.gov/oce/climate_change/techguide/Denef_et_al_2011_Review_of_reviews_v1.0.pdf. (20 May 2016).

DeWalle, D.R.; Heisler, G.M. 1988. Use of windbreaks for home energy conservation. Agriculture, Ecosystems & Environment. 22/23: 243–260.

Dixon, R.K.; Winjum, J.K.; Andrasko, K.J. [et al.]. 1994. Integrated land-use systems: assessment of promising agroforest and alternative land-use practices to enhance carbon conservation and sequestration. Climatic Change. 27(1): 71–92.

Dube, F.; Thevathasan, N.V.; Zagal, E. [et al.]. 2011. Carbon sequestration potential of silvopastoral and other land use systems in the Chilean Patagonia. In: Kumar, B.M.; Nair, P.K.R., eds. Carbon sequestration potential of agroforestry systems: opportunities and challenges. Advances in Agroforestry. Vol. 8. New York: Springer: 101–127.

Duguma, L.A.; Minang, P.A.; van Noordwijk, M. 2014. Climate change mitigation and adaptation in the land use sector: from complementarity to synergy. Environmental Management. 54(3): 420–432.

Dutaur, L.; Verchot, L.V. 2007. A global inventory of the soil CH_4 sink. Global Biogeochemical Cycles. 21: GB4013. doi:10.1029/2006GB002734. http://onlinelibrary.wiley.com/doi/10.1029/2006GB002734/pdf. (20 May 2016).

Eagle, A.J.; Olander, L.P. 2012. Greenhouse gas mitigation with agricultural land management activities in the United States: a side-by-side comparison of biophysical potential. Advances in Agronomy. Vol. 115. San Diego, CA: Elsevier: 81–179.

Eve, M.; Pape, D.; Flugge, M. [et al.], eds. 2014. Quantifying greenhouse gas fluxes in agriculture and forestry: methods for entity-scale inventory. Tech. Bull. No. 1939. Washington, DC: U.S. Department of Agriculture, Office of the Chief Economist. 606 p. http://www.usda.gov/oce/climate_change/Quantifying_GHG/USDATB1939_07072014.pdf. (20 May 2016).

Evers, A.K.; Bambrick, A.; Lacombe, S. [et al.]. 2010. Potential greenhouse gas mitigation through temperate tree-based intercropping systems. The Open Agriculture Journal. 4(4): 49–57.

Fry, J.A.; Xian, G.; Jin, S. [et al.]. 2011. Completion of the 2006 National Land Cover Database for the conterminous United States. Photogrammetric Engineering and Remote Sensing. 77(9): 858–864.

Gorte, R.W. 2009. U.S. tree planting for carbon sequestration. CRS Report for Congress R40562. Washington, DC: Congressional Research Service: 5 p.

Grandy, A.S.; Salam, D.S.; Wickings, K. [et al.]. 2013. Soil respiration and litter decomposition responses to nitrogen fertilization rate in no-till corn systems. Agriculture, Ecosystems & Environment. 179: 35–40.

Groffman, P.M.; Gold, A.J.; Addy, K. 2000. Nitrous oxide production in riparian zones and its importance to national emission inventories. Chemosphere-Global Change Science. 2(3): 291–299.

Haile, S.G.; Nair, P.K.R.; Nair, V.D. 2008. Carbon storage of different soil-size fractions in Florida silvopasture systems. Journal of Environmental Quality. 37(5): 1789–1797.

Haile, S.G.; Nair, V.D.; Nair, P.K.R. 2010. Contribution of trees to carbon storage in soils of silvopasture systems in Florida, USA. Global Change Biology. 16(1): 427–438.

Harmon, M.E.; Woodall, C.W.; Fasth, B. [et al.]. 2011. Differences between standing and downed dead tree wood density reduction factors: a comparison across decay classes and tree species. Res. Pap. NRS-15. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 40 p.

Hernandez-Ramirez, G.; Sauer, T.J.; Cambardella, C.A. [et al.]. 2011. Carbon sources and dynamics in afforested and cultivated corn belt soils. Soil Science Society of America Journal. 75(1): 216–225.

Homer, C.; Huang, C.; Yang, L. [et al.]. 2004. Development of a 2001 National Land Cover Database for the United States. Photogrammetric Engineering and Remote Sensing. 44(7): 3999–4005.

Hooker, T.D.; Compton, J.E. 2003. Forest ecosystem carbon and nitrogen accumulation during the first century after agricultural abandonment. Ecological Applications. 3(2): 299–313.

Hoover, C.; Birdsey, R.C.; Goines, B. [et al.]. 2014. Quantifying greenhouse gas sources and sinks in managed forest systems. In: Eve, M.; Pape, D.; Flugge, M. et al., eds. Quantifying greenhouse gas fluxes in agriculture and forestry: methods for entity-scale inventory. Tech. Bull. No. 1939. Washington, DC: U.S. Department of Agriculture, Office of the Chief Economist. 606 p. Chap. 6. http://usda.gov/oce/climate_change/Quantifying_GHG/Chapter6S.pdf. (20 May 2014).

Howlett, D.S.; Mosquera-Losada, M.R.; Nair, P.K.R. 2011. Soil carbon storage in silvopastoral systems and a treeless pasture in northwestern Spain. Journal of Environmental Quality. 40(3): 825–832.

ICF International (ICF). 2013. Greenhouse gas mitigation options and costs for agricultural land and animal production within the United States. Prepared for the U.S. Department of Agriculture, Climate Change Program Office. Washington, DC: ICF International. 270 p.

Intergovernmental Panel on Climate Change (IPCC). 2000. Good practice guidance and uncertainty management in national greenhouse gas inventories. http://www.ipcc-nggip.iges.or.jp/public/gp/english/. (15 January 2015).

Intergovernmental Panel on Climate Change (IPCC). 2006. Guidelines for national greenhouse gas inventories. Prepared by the National Greenhouse Gas Inventories Programme: Eggleston, H.S.; Buendia, L.; Miwa, K. [et al.]. Hayama, Japan: Institute for Global Environmental Strategies. http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html. (20 May 2016).

Janssens, I.A.; Dieleman, W.; Luyssaert, S. [et al.]. 2010. Reduction of forest soil respiration in response to nitrogen deposition. Nature Geoscience. 3(5): 315–322.

Jenkins, J.C.; Chojnacky, D.C.; Heath, L.S.; Birdsey, R.A. 2003. National-scale biomass estimators for United States tree species. Forest Science. 49(1): 12–35.

Kim, D.-G.; Isenhart, T.M.; Parkin, T.B. [et al.]. 2009. Nitrate and dissolved nitrous oxide in groundwater within cropped fields and riparian buffers. Biogeosciences Discussions. 6(1): 651–685.

Kim, D.-G.; Isenhart, T.M.; Parkin, T.B. [et al.]. 2010. Methane flux in cropland and adjacent riparian buffers with different vegetation covers. Journal of Environmental Quality. 39(1): 97–105.

Kort, J.; Turnock, R. 1999. Carbon reservoir and biomass in Canadian prairie shelterbelts. Agroforestry Systems. 44(2–3): 175–186.

Kumar, B.M.; Nair, P.K.R., eds. 2011. Carbon sequestration potential of agroforestry systems: opportunities and challenges. Advances in Agroforestry. Vol. 8. Dordrecht, Netherlands: Springer. 307 p.

Kursten, E.; Burschel, P. 1993. CO_2 mitigation by agroforestry. Water Air Soil Pollution. 70(1–4): 533–544.

Lal, R. 2004. Carbon emission from farm operations. Environment International. 30(7): 981–990.

Lewandrowski, J.; Peters, M.; Jones, C. [et al.]. 2004. Economics of sequestering carbon in the U.S. agricultural sector. Tech. Bull. TB-1909. Washington, DC: U.S. Department of Agriculture, Economic Research Service. http://www.ers.usda.gov/publications/tb-technical-bulletin/tb1909.aspx. (20 May 2016).

Liebig, M.A.; Franzluebbers, A.J.; Follett, R.F., eds. 2012. Managing agricultural greenhouse gases: coordinated agricultural research through GRACEnet to address our changing climate. San Diego, CA: Academic Press. 576 p.

Liknes, G.C.; Perry, C.H.; Meneguzzo, D.M. 2010. Assessing tree cover in agricultural landscapes using high-resolution aerial imagery. Journal of Terrestrial Observation. 2(1): 38–55.

Lin, C.H.; McGraw, R.L.; George, M.F.; Garrett, H.E. 1998. Shade effects on forage crops with potential in temperate agroforestry practices. Agroforestry Systems. 44(2–3): 109–119.

López-Díaz, M.L.; Rolo, V.; Moreno, G. 2011. Trees' role in nitrogen leaching after organic, mineral fertilization: a greenhouse experiment. Journal of Environmental Quality. 40(3): 853–859.

McCarty, G.W.; Ritchie, J.C. 2002. Impact of soil movement on carbon sequestration in agricultural ecosystems. Environmental Pollution. 116(3): 423–430.

McKinley, D.C.; Ryan, M.G.; Birdsey, R.A. [et al.]. 2011. A synthesis of current knowledge on forests and carbon storage in the United States. Ecological Applications. 21(6): 1902–1924.

Meneguzzo, D.M.; Liknes, G.C.; Nelson, M.D. 2013. Mapping trees outside forests using high-resolution aerial imagery: a comparison of pixel- and object-based classification approaches. Environmental Monitoring and Assessment. 185(8): 6261–6275.

Mitlöhner, F.M.; Morrow, J.L.; Dailey, J.W. [et al.]. 2001. Shade and water misting effects on behavior, physiology, performance, and carcass traits of heat-stressed feedlot cattle. Journal of Animal Science. 79(1): 2327–2335.

Motocha, J.; Schroth, G.; Hills, T.; Hole, D. 2012. Integrating climate change adaptation and mitigation through agroforestry and ecosystem conservation. In: Nair, P.K.R.; Garrity, D.P., eds. Agroforestry: the future of global land use. Advances in Agroforestry. Vol. 9. New York: Springer: 105–126.

Nair, P.K.R. 2012a. Climate change mitigation and adaptation: a low hanging fruit of agroforestry. In: Nair, P.K.R.; Garrity, D.P., eds. Agroforestry: the future of global land use. Advances in Agroforestry. Vol. 9. New York: Springer: 31–67.

Nair, P.K.R. 2012b. Carbon sequestration studies in agroforestry systems: a reality-check. Agroforestry Systems. 86(2): 243–253.

Nair, P.K.R.; Kumar, B.M.; Nair, V.D. 2009. Agroforestry as a strategy for carbon sequestration. Journal of Plant Nutrition and Soil Science. 172(1): 10–23.

Nair, P.K.R.; Nair, V.D. 2003. Carbon storage in North American agroforestry systems. In: Kimble, J.; Heath, L.; Birdsey, R.; Lal, R., eds. The potential of U.S. forest soils to sequester carbon and mitigate the greenhouse effect. Boca Raton, FL: CRC Press: 333–346.

Nair, P.K.R.; Nair, V.D.; Kumar, B.M.; Showalter, J.M. 2010. Carbon sequestration in agroforestry systems. In: Sparks, D., ed. Advances in Agronomy. Vol. 108. San Diego, CA: Academic Press: 237–307.

Nair, V.D.; Haile, S.G.; Michel, G.A.; Nair, P.K. 2007. Environmental quality improvement of agricultural lands through silvopasture in southeastern United States. Scientia Agricola. 64(5): 513–519.

Nave, L.E.; Swanston, C.W.; Mishra, U.; Nadelhoffer, K.J. 2013. Afforestation effects on soil carbon storage in the United States: a synthesis. Soil Science Society of America Journal. 77(3): 1035–1047.

Nuberg, I.K. 1998. Effect of shelter on temperate crops: a review to define research for Australian conditions. Agroforestry Systems. 41(1): 3–34.

Niu, X.; Duiker, S.W. 2006. Carbon sequestration potential by afforestation of marginal agricultural land in the Midwestern U.S. Forest Ecology and Management. 223(1): 415–427.

Ogle, S.M.; Adler, P.R.; Breidt, F.J. [et al.]. 2014. Quantifying greenhouse gas sources and sinks in cropland and grazing land systems. In: Eve, M.; Pape, D.; Flugge, M. [et al.], eds. Quantifying greenhouse gas fluxes in agriculture and forestry: methods for entity-scale inventory. Tech. Bull. No. 1939. Washington, DC: U.S. Department of Agriculture, Office of the Chief Economist. 606 p. Chap. 3. http://www.usda.gov/oce/climate_change/Quantifying_GHG/Chapter3S.pdf. (20 May 2016).

Ogle, S.M.; Breidt, F.; Easter, M. [et al.]. 2010. Scale and uncertainty in modeled soil organic carbon stock changes for US croplands using a process-based model. Global Change Biology. 16(2): 810–822.

Ogle, S.M.; Eve, M.D.; Breidt, F.J.; Paustian, K. 2003. Uncertainty in estimating land use and management impacts on soil organic carbon storage for U.S. agroecosystems between 1982 and 1997. Global Change Biology. 9(11): 1521–1542.

Olander, L.; Wollenberg, E.; Tubiello, F.; Herold, M. 2013. Advancing agricultural greenhouse gas quantification. Environmental Research Letters. 8: 011002. doi:1088/1748-9326/8/1/011002. http://iopscience.iop.org/1748-9326/8/1/011002. (20 May 2016).

Olson, R.; Schoeneberger, M.; Aschmann, S. 2000. An ecological foundation for temperate agroforestry. In: Garrett, H.E.; Rietveld, W.J.; Fisher, R., eds. North American agroforestry: an integrated science and practice. ASA Special Publication. Madison, WI: American Society of Agronomy: 31–61.

Parton, W.J.; Hartman, M.D.; Ojima, D.S.; Schimel, D.S. 1998. DAYCENT: its land surface submodel: description and testing. Global and Planetary Change. 19(1): 35–48.

Parton, W.J.; Schimel, D.S.; Cole, C.V.; Ojima, D.S. 1987. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. Soil Science Society of America Journal. 51(5): 1173–1179.

Paul, E.A.; Morris, S.J.; Six, J. [et al.]. 2003. Interpretation of soil carbon and nitrogen dynamics in agricultural and afforested soils. Soil Science Society of America Journal. 67(5): 1620–1628.

Pearson, T.R.H.; Brown, S.L.; Birdsey, R.A. 2007. Measurement guidelines for the sequestration of forest carbon. Gen. Tech. Rep. NRS-18. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 42 p.

Peichl, M.; Thevathasan, N.V.; Gordon, A.M. [et al.]. 2006. Carbon sequestration potentials in temperate tree-based intercropping systems, southern Ontario, Canada. Agroforestry Systems. 66(3): 243–257.

Perry, C.H.; Woodall, C.W.; Liknes, G.C.; Schoeneberger, M.M. 2009. Filling the gap: improving estimates of working tree resources in agricultural landscapes. Agroforestry Systems. 75(1): 91–101.

Perry, C.H.; Woodall, C.W.; Schoeneberger, M.M. 2005. Inventorying trees in agricultural landscapes: toward an accounting of working trees. In: Brooks, K.N.; Folliott, P.F., eds. Moving agroforestry into the mainstream. Proceedings of the 9th North American agroforestry conference, Rochester, MN. St. Paul, MN: University of Minnesota, Department of Forest Resources: 12–15. http://www.srs.fs.usda.gov/pubs/ja/ja_perry008.pdf. (20 May 2016).

Plieninger, T. 2011. Capitalizing on the carbon sequestration potential of agroforestry in Germany's agricultural landscapes: realigning the climate mitigation and landscape conservation agendas. Landscape Research. 36(4): 435–454.

Possu, W.B. 2015. Carbon storage potential of windbreaks on agricultural lands of the continental United States. Dissertation, University of Nebraska-Lincoln. 229 p. http://search.proquest.com/docview/1710737772?accountid=281477. (8 November 2016).

Priano, M.E.; Fuse, V.S.; Gere, J.I. [et al.]. 2014. Tree plantations on a grassland region: effects on methane uptake by soils. Agroforestry Systems. 88(1): 187–191.

Ramirez, K.S.; Craine, J.M.; Fierer, N. 2012. Consistent effects of nitrogen amendments on soil microbial communities and processes across biomes. Global Change Biology: 18(6): 1918–1927.

Ritson, P.; Sochacki, S. 2003. Measurement and prediction of biomass and carbon content of *Pinus pinaster* trees in farm forestry plantations, south-western Australia. Forest Ecology and Management: 175(1): 103–117.

Robertson, G.; Mason, A., eds. 2016. Assessing the sustainability of agricultural and urban forests in the United States. FS-1067. Washington, DC: U.S. Department of Agriculture, Forest Service. 75 p.

Russell, M.B.; D'Amato, A.W.; Schulz, B. [et al.]. 2014. Quantifying understory vegetation in the US Lake States: a proposed framework to inform regional forest carbon stocks. Forestry. 87(5): 629–638.

Ryszkowski, L.; Kedziora, A. 2007. Modification of water flows and nitrogen fluxes by shelterbelts. Ecological Engineering. 29(4): 388–400.

Sauer, T.J.; Cambardella, C.A.; Brandle, J.R. 2007. Soil carbon and tree litter dynamics in a red cedar-scotch pine shelterbelt. Agroforestry Systems. 71(3): 163–174.

Schnell, S.; Altrell, D.; Ståhl, G.; Kleinn, C. 2015. The contribution of trees outside forests to national tree biomass and carbon stocks—a comparative study across three continents. Environmental Monitoring and Assessment. 187(1): 1–18.

Schoeneberger, M.; Bentrup, G.; de Gooijer, H. [et al.]. 2012. Branching out: agroforestry as a climate change mitigation and adaptation tool for agriculture. Journal of Soil and Water Conservation. 67(5): 128A–136A.

Schoeneberger, M.M. 2009. Agroforestry: working trees for sequestering carbon on agricultural lands. Agroforestry Systems. 75(1): 27–37.

Schoeneberger, M.M.; Bentrup, G.; Current, D. [et al.]. 2008. Building bigger better buffers for bioenergy. Water Resources Impact. 10: 22–26.

Sharrow, S.H.; Ismail, S. 2004. Carbon and nitrogen storage in agroforests, tree plantations, and pastures in western Oregon, USA. Agroforestry Systems. 60(2): 123–130.

Smith, J.E.; Heath, L.S.; Skog, K.E.; Birdsey, R.A. 2006. Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States. Gen. Tech. Rep. NE-343. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station. 216 p.

Sudmeyer, R.A.; Scott, P.R. 2002. Characterisation of a windbreak system on the south coast of western Australia. 1. Microclimate and wind erosion. Animal Production Science. 42(6): 703–715.

Suwanwaree, P.; Robertson, G.P. 2005. Methane oxidation in forest, successional, and no-till agricultural ecosystems. Soil Science Society of America Journal. 69(6): 1722–1729.

Topp, E.; Pattey, E. 1997. Soils as sources and sinks for atmospheric methane. Canadian Journal of Soil Science. 77(2): 167–177.

Tufekcioglu, A.; Reich, J.W.; Isenhart, T.M.; Schultz, R.C. 2003. Biomass, carbon and nitrogen dynamics of multi-species riparian buffers within an agricultural watershed in Iowa, USA. Agroforestry Systems. 57(3): 187–198.

Udawatta, R.P.; Jose, S. 2012. Agroforestry strategies to sequester carbon in temperate North America. Agroforestry Systems. 86(2): 225–242.

Udawatta, R.P.; Kremer, R.J.; Garrett, H.E.; Anderson, S.H. 2009. Soil enzyme activities and physical properties in a water-shed managed under agroforestry and row-crop systems. Agriculture, Ecosystems and Environment. 131(1): 98–104.

U.S. Department of Agriculture (USDA), Forest Service. 2013. FIA database description and user's manual for phase 2, ver. 6.0.1. http://www.fia.fs.fed.us/library/database-documentation/current/ver60/FIADB%20User%20Guide%20P2_6-0-2_final-opt.pdf. (20 May 2016).

U.S. Department of Agriculture, Natural Resources Conservation Service (USDA-NRCS). 2009. Summary report: 2007 national resources inventory. Washington, DC: U.S. Department of Agriculture, Natural Resources Conservation Service; Ames, IA: Iowa State University, Center for Survey Statistics and Methodology. http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1041379.pdf. (20 May 2016).

U.S. Department of Agriculture, Office of the Chief Economist (USDA-OCE). 2016. U.S. agriculture and forestry greenhouse gas inventory: 1990–2013. Tech. Bull. No. 1943. Washington, DC: United States Department of Agriculture, Office of the Chief Economist, Climate Change Program Office. 137 p. http://www.usda.gov/oce/climate_change/AFGG_Inventory/USDA_GHG_Inventory_1990-2013_9_19_16_reduced.pdf. (22 October 2016).

U.S. Environmental Protection Agency (EPA). 2014. Inventory of U.S. greenhouse gas emissions and sinks: 1990–2012. EPA 430-R-14-003. Washington, DC: U.S. Environmental Protection Agency. http://www.epa.gov/climatechange/Downloads/ghgemissions/US-GHG-Inventory-2014-Main-Text.pdf. (20 May 2016).

Verchot, L.V.; Van Noordwijk, M.; Kandji, S. [et al.]. 2007. Climate change: linking adaptation and mitigation through agroforestry. Mitigation and Adaptation Strategies for Global Change. 12(5): 901–918.

Vesterdal, L.; Ritter, E.L.; Gunderson, P. 2002. Change in soil organic carbon following afforestation of former arable land. Forest Ecology and Management. 169(1): 137–147.

Vira, B.; Agarwal B.; Jamnadass, R.H. [et al.] 2015. Forests, trees and landscapes for food security and nutrition. In: Vira, B.; Wildburger, C.; Mansourian, S., eds. Forests and food: addressing hunger and nutrition across sustainable landscapes. Cambridge, United Kingdom: Open Book Publishers: 9–28.

Wallace, D.C.; Geyer, W.A.; Dwyer, J.P. 2000. Waterbreaks: managed trees for the floodplain. Agroforestry Notes: Special Applications #4. http://nac.unl.edu/documents/agroforestry-notes/an19sa04.pdf. (20 May 2016).

Walthall, C.; Hatfield, J.; Backlund, P. [et al.]. 2012. Climate change and agriculture in the United States: effects and adaptation. USDA Tech. Bull. No. 1935. Washington, DC: U.S. Department of Agriculture. 185 p.

West, T.O.; Bandaru, V.; Brandt, C.C.; Schuh, A.E. [et al.]. 2011. Regional uptake and release of crop carbon in the United States. Biogeosciences. 8(8): 631–654.

Woodall, C.W.; Heath, L.S.; Domke, G.M.; Nichols, M. 2011. Methods and equations for estimating aboveground volume, biomass, and carbon for forest trees in the U.S.'s national inventory, 2010. Gen. Tech. Rep. NRS-88. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 30 p.

Wotherspoon, A.; Thevathasan, N.V.; Gordon, A.M.; Voroney, R.P. 2014. Carbon sequestration potential of five tree species in a 25-year-old temperate tree-based intercropping system in southern Ontario, Canada. Agroforestry Systems. 88(4): 631–643.

Zhou, X. 1999. On the three-dimensional aerodynamic structure of shelterbelts. Lincoln, NE: University of Nebraska. 197 p. Ph.D. dissertation.

Zhou, X.; Schoeneberger, M.; Brandle, J. [et al.]. 2015. Analyzing the uncertainties in use of forest-derived biomass equations for open-grown trees in agricultural land. Forest Science. 61(1): 144–161.

Zhou, X.H.; Brandle, J.R.; Awada, T.N. [et al.]. 2011. The use of forest-derived specific gravity for the conversion of volume to biomass for open-grown trees on agricultural land. Biomass and Bioenergy. 35(5): 1721–1731.

Zomer, R.J.; Neufeldt, H.; Xu, J. [et al.]. 2016. Global tree cover and biomass carbon on agricultural land: the contribution of agroforestry to global and national carbon budgets. Scientific Reports 6: 29987. doi:10.1038/srep29987. http://www.nature.com/articles/srep29987. (22 October 2016).