Chapter 15
Carbon Dynamics Following the Creation of Early Successional Habitats in Forests of the Central Hardwood Region

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Abstract Across a forested landscape, stand-level management actions or natural disturbances that create early successional habitats result in a short-term loss of carbon in any given stand, but are often offset by carbon gains in other, undisturbed stands. Standing carbon stocks and rates of sequestration vary with species, site productivity, stand age, and stand structure. The age distribution of forest stands has a particularly large effect on landscape-level carbon storage. Consequently, forest management activities, including creation of early successional habitats, have short-term implications for stand-level carbon storage, but their impact on forest- or landscape-level carbon storage ultimately depends upon the temporal distribution and spatial scale of young forest stands on the landscape.

15.1 Introduction

Anthropogenic activities, including burning fossil fuels and changes in land-use patterns, have increased atmospheric concentrations of greenhouse gases. Considered to be the most important anthropogenic greenhouse gas, atmospheric carbon dioxide (CO_2) concentration reached 387 ppm in 2009, significantly higher than the pre-industrial concentration estimated at 280 ppm (http://www.esrl.noaa.gov/gmd/ccggl/trends). Future emission scenarios suggest CO_2 concentrations will increase between 41 and 158% by 2100 (IPCC 2007).

The rise in CO_2 and other greenhouse gases have led to a 0.7 °C increase in global surface temperature during the twentieth century (IPCC 2007). Given current and

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predicted CO_2 levels over the next 100 years, future changes in global surface temperatures are expected to be even larger. Under all emission scenarios modeled by the Intergovernmental Panel on Climate Change (IPCC), fossil fuel use and CO_2 emissions will increase well into the 21st century. Mitigation efforts designed to capture and store carbon, however, can offset and regulate anthropogenic CO_2 emissions. The ability of forestland to sequester and offset CO_2 emissions has generated substantial interest in managing forests for increased CO_2 uptake and storage through activities such as afforestation, reforestation, and improved forest management.

Forests sequester atmospheric carbon in aboveground live and dead biomass, soil organic matter, roots, and surface detritus. They emit carbon back to the atmosphere through metabolic processes such as autotrophic and heterotrophic respiration, and physical processes such as herbivory, fire, insect and disease outbreaks, and timber harvesting. The forest carbon cycle is comprised of periods of carbon storage punctuated by periodic disturbance events (both natural and anthropogenic) that release carbon back to the atmosphere. At any given point in time, individual stands are either sources or sinks of carbon. This difference between gross carbon gained via photosynthesis, or gross primary productivity (GPP), and total carbon lost is net ecosystem production (NEP). When NEP is negative, a forest stand is a source of atmospheric carbon (Fig. 15.1). When NEP is positive, a forest stand is removing and storing carbon. Although individual stands are either sources or sinks of carbon at any specific point in time, carbon balance at the forest- or landscape-level is determined by summing the net carbon balance of individual stands.

At a global scale, forest ecosystems are a net carbon sink; over a defined timeframe, forests sequester more carbon than they emit. Forestland in the USA constitutes a particularly large carbon sink relative to the global carbon budget. Overall, forestland in the USA sequestered 216 Tg carbon in 2008, an 18% increase over that

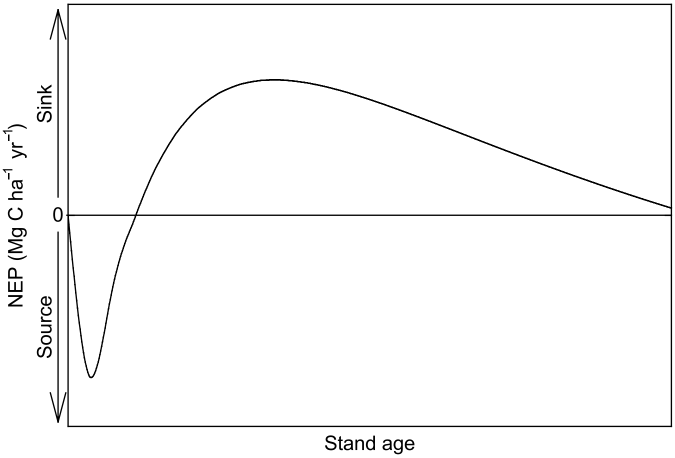

Fig. 15.1 Generalized pattern of net ecosystem production (NEP) as a function of stand age following stand replacing disturbance (e.g., regeneration harvest)

in 1990 (Environmental Protection Agency (EPA) 430-R-10-006). Much of this increase is the result of improved forest management practices, successful forest regeneration treatments, and afforestation and reforestation efforts, along with increases in aboveground live biomass in existing forest stands.

For purposes of inventory and accounting, carbon in forests is categorized into one of six pools: (1) aboveground biomass (all living biomass above the soil); (2) belowground biomass (all living biomass of roots >2 mm in diameter); (3) dead wood (non-living woody biomass); (4) litter [litter (O_i) and duff ($O_e + O_a$) layers in addition to woody material <7.5 cm]; (5) soil organic matter; and (6) harvested wood products (IPCC 2003, 2006). Because carbon is not directly measured as part of standard forest inventory procedures, methods exist to estimate carbon storage in the various carbon pools using forest inventory data. For example, aboveground live tree carbon is often estimated using species-specific allometric equations that relate tree size to biomass. Carbon is then calculated using a ratio that relates biomass (on a dry weight basis) to carbon [0.5 for aboveground live tree biomass (IPCC 2003)]. This biomass to carbon conversion factor varies based on the carbon pool analyzed [e.g., litter, dead wood, etc. (IPCC 2003)]. Excluding the harvested wood product carbon pool, which the size and longevity will vary with silvicultural prescription, harvesting system, species, and forest product type (Smith et al. 2006), aboveground biomass and mineral soil carbon pools constitute the greatest proportion of a forest stand's overall carbon stock; these are followed by belowground biomass, dead wood, and litter carbon pools.

Predominant forest types within the Central Hardwood Region are dominated by upland oak and hickory (*Quercus-Carya*) species (Johnson et al. 2002b). Associated canopy-tree species include mixed-mesophytic species such as yellow-poplar (*Liriodendron tulipifera*), ashes (*Fraxinus* spp.), maples (*Acer spp.*), and others (see Loftis et al., Chap. 5). As abandoned farmland returns to forestland and past timber harvests enter mid- to later stages of stand development, carbon continues to accumulate in these forests, making carbon capture and storage an added benefit of past management actions. Relative to other eastern forest types, second growth oak-hickory forests are a strong carbon sink; they are capable of sequestering 5.25 Mg carbon ha^{-1} $year^{-1}$ (Greco and Baldocchi 1996) compared to 3.7 Mg carbon ha^{-1} $year^{-1}$ in northern hardwood forests in the northeastern USA (Wofsy et al. 1993).

Upland hardwood forests of the Central Hardwood Region are managed for a multitude of ecosystem services. While meeting specific resource objectives, the creation of early successional habitats in mature upland hardwood forests can alter stand-level carbon storage and, depending on the temporal and spatial scale of management actions, landscape-level carbon dynamics (e.g., Campbell et al. 2004; Depro et al. 2008). Variations in climate, forest type, stand structure and species composition, stand age, and edaphic conditions among and within the physiographic regions of the central hardwoods interact to influence the rates of carbon sequestration and total carbon storage following both natural and silvicultural disturbance events. Although forest carbon storage is most meaningful when examined at the landscape-level (Harmon 2001), silvicultural prescriptions designed to achieve resource management objectives are implemented at the stand-level. Consequently, developing an understanding of the role forest management has on the various carbon pools and stand-level carbon storage is relevant to carbon management at the landscape-level. The following provides a synopsis of the literature associated with the potential effects of creating early successional habitat via silvicultural practices on stand-level biological carbon pools and dynamics in the Central Hardwood Region.

15.2 Carbon Dynamics Following the Creation of Early Successional Habitats

15.2.1 Aboveground Biomass

As described by Loftis et al. (Chap. 5), numerous silvicultural tools and methods can be used to create early successional habitats. Even-aged silviculture utilizing the clearcutting and shelterwood methods, a two-aged system or shelterwood with reserves, and the group selection method of uneven-aged management are all effective methods. In all cases, these regeneration harvests result in stands that lack closed or continuous canopy and are in the early stages of stand development (i.e., stand initiation).

Of the biological carbon pools after harvest (i.e., excluding the harvested carbon pool), carbon stored in aboveground biomass is the most dynamic as well as most easily quantified (Fahey et al. 2010). Following harvests and barring any managed or unmanaged disturbance events, the aboveground component of a newly regenerated forest stand accumulates carbon in a predictable sigmoid pattern (Hunt 1982) (Fig. 15.2). Young forests take up carbon rapidly and have high growth rates, but

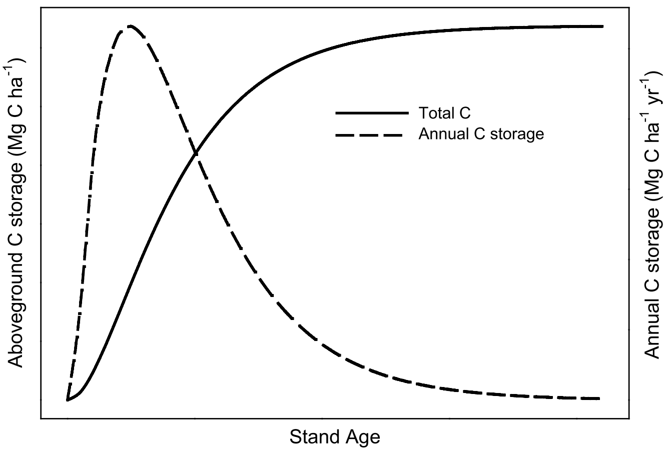

Fig. 15.2 Generic pattern of stand-level total carbon (C) storage and annual rate of C storage in the aboveground biomass pool following a regeneration harvest as a function of stand age

contain significantly less biomass and standing carbon stocks than mature forest stands. During canopy closure, net primary productivity and carbon uptake by aboveground vegetation are maximized. As stands age, rates of carbon sequestration and net primary productivity decline, but aboveground biomass and carbon storage continue to increase until biomass approaches a maximum defined, in part, by species composition and associated maximum size-density relations (Yoda et al. 1963) and environmental conditions (Gholz 1982; Johnson et al. 2000; Kranabetter 2009).

Carbon stored in the aboveground biomass pool is the largest (Li et al. 2007) or second largest (Bolstad and Vose 2005) carbon pool in upland forests of the Central Hardwood Region and is maximized when the frequency of anthropogenic and natural disturbances is low (Reinhardt and Holsinger 2010; Nunery and Keeton 2010; Harmon et al. 2009; Hudiburg et al. 2009; Harmon and Marks 2002; Janisch and Harmon 2002). The pattern of carbon accumulation observed throughout stand development (Fig.15.2) is applicable to forest stands regardless of species composition (Spetich et al. 1998, Johnsen et al. 2001; Law et al. 2003; Taylor et al. 2007). However, the upper limit of forest stand-level carbon storage and the timeframe associated with maximizing carbon storage in aboveground biomass differ among forest stands due to variation in species composition (Grigal and Ohmann 1992; Caspersen and Pacala 2001; Bunker et al. 2005), disturbance patterns, and edapho-climatic conditions (Smithwick et al. 2002; Van Tuyl et al. 2005; Hudiburg et al. 2009). As a result of their history (see White et al., Chap. 3; Shifley and Thompson, Chap. 6), most regenerated upland forests in the Central Hardwood Region, are even-aged, second-growth forests (80–100 years old) and are at a point in stand development where they are still accumulating aboveground carbon (Brown et al. 1997; Brown and Schroeder 1999) albeit at a slower rate than occurred earlier in stand development (i.e., during stand initiation; Fig. 15.2).

Although second growth upland hardwood forests throughout the Central Hardwood Region are still accumulating carbon in aboveground biomass, aboveground carbon stocks are substantially less than in comparable old-growth forests. Throughout oak-hickory dominated regions, aboveground carbon stocks (calculated using a standard carbon to biomass ratio of 0.5) can range between 87.5 and 92.5 Mg carbon ha^{-1} for mature, sawtimber sized stands and 50–62.5 Mg carbon ha^{-1} for poletimber sized stands (Brown et al. 1997). Stands recently regenerated (i.e., seedling/sapling sized stands with early successional and young forest structure) generally store ≤25 Mg carbon ha^{-1} in aboveground live biomass (Brown et al. 1997). However, these carbon estimates reflect averages for oak-hickory forests across a broad range of physiographic regions and age classes. Aboveground carbon stocks vary with site productivity and species composition. In Indiana, for instance, aboveground carbon stocks in oak-hickory forests vary by as much as 50% between stands of low [oak site index (SI), base-age 50 between 16.8 and 18.3 m] and high (oak SI between 25.9 and 27.4 m) site quality (Kaczmarek et al. 1995).

If undisturbed, upland hardwood forests have the potential to sequester substantial quantities of carbon in aboveground vegetation. Old-growth forests across the Central Hardwood Region (Schmelz and Lindsey 1965; McClain and Ebinger 1968; Weaver and Ashby 1971; Muller 1982; Cho and Boerner 1991; Spetich and Parker

1998) possess aboveground carbon stocks that range from 98.5 Mg carbon ha^{-1} in oak-hickory forests in Illinois (McClain and Ebinger 1968) to 165 Mg carbon ha^{-1} in mixed-mesophytic forests of Kentucky (Muller 1982). As edaphic and climatic (i.e., edaphoclimatic) conditions improve, the capacity of a forest stand to accumulate and store aboveground biomass and carbon increases (Kranabetter 2009). A prime example of this positive productivity-carbon storage relationship is found in forests of Great Smoky Mountains National Park. Here, the edophoclimatic environment, species composition, and disturbance history have resulted in a substantial and prolonged accumulation of biomass with aboveground live tree carbon stocks estimated to range from 163 to197 Mg carbon ha^{-1} in some of the park's old-growth, cove hardwood forests (Busing et al. 1993). Old growth forests are a significant carbon sink (Luyssaert et al. 2008), but occupy only a small percentage of forested land relative to second-growth forests. Old growth forests represent the upper bounds of carbon storage that may not be attainable for the majority of second-growth forest stands, even if left unmanaged (Brown et al. 1997).

15.2.2 Belowground Biomass

The belowground biomass pool is composed of coarse and fine roots (IPCC 2003). Despite consuming a significant proportion of annual net primary productivity, fine roots contribute only a minor proportion of the total carbon storage in any given forest stand (Santantonio et al. 1977; Grier et al. 1981; Comeau and Kimmins 1989; John et al. 2001). Carbon is stored in coarse roots in two distinct pools: living vegetation and harvested or dead trees (hereafter referred to as residual coarse roots). Unlike fine roots, coarse roots represent a significant and long-term carbon pool (Resh et al. 2003; Miller et al. 2006; Yanai et al. 2006). Similar to aboveground biomass, carbon storage in coarse roots accumulates in a sigmoid pattern as stands progress through stages of stand development. In young, recently regenerated stands, small diameter trees require less structural support than large trees of older stands. Consequently, coarse root biomass and corresponding carbon stocks are relatively low early in stand development and are maximized during the later stages of stand development (Misra et al. 1998; Resh et al. 2003; Yanai et al. 2006).

Following a regeneration harvest, coarse roots of harvested trees remain on-site, and the carbon stored in them may have a relatively long residence time. The rate of decomposition of residual coarse root systems depends on tree species and wood quality (Fahey and Arthur 1994; Chen et al. 2001), size of residual coarse root material (Fahey et al. 1988; Janisch et al. 2005), and temperature and moisture availability (Fahey and Arthur 1994; Ximenes and Gardner 2006). Decomposition of residual coarse root systems is often modeled using a negative exponential function (Melin et al. 2009), suggesting that a portion of residual coarse root carbon is a long-term pool that must be accounted for when assessing the effects of management on stand-level carbon storage. In loblolly pine (*Pinus taeda*), for example, coarse- and tap-root biomass can persist upwards of 60 years after harvest (Ludovici et al. 2002) and between 10% and 50% of hardwood coarse root biomass can remain on-site 100 years following harvest in Australia (Ximenes and Gardner 2006).

Information regarding coarse root decomposition specific to upland hardwood tree species in the Central Hardwood Region is sparse. However, it is plausible that carbon stored in coarse roots systems following harvests in hardwood stands may be of greater importance to a stand's overall carbon budget than in conifer-dominated stands, given the propensity of hardwood species to allocate more carbon belowground than managed conifer species (e.g., Miller et al. 2006). As a guideline, the IPCC suggests that carbon stored in the biomass of coarse root systems has a post-harvest residence time of only 10 years (IPCC 2003). From the limited data available on coarse root decomposition, (e.g., Ludovici et al. 2002; Ximenes and Gardner 2006) it would appear this 10-year residence time (IPCC 2003) underestimates carbon storage in residual root systems. In turn, this overestimation of residual root decay can lead to an overestimate of carbon flux to the atmosphere following forest management and underestimate stand-level carbon stocks over time.

15.2.3 Dead Wood Biomass

Dead and down coarse wood [defined as dead and down woody biomass greater than a specific diameter (often >7.6 cm in diameter or >10 cm in diameter)] is vital to proper maintenance of ecosystem structure and function. Dead and down wood fills an important niche in forested ecosystems, providing habitat for vertebrate species (e.g., Moorman et al., Chap. 11) as well as an energy and nutrient substrate for invertebrate, fungal, and microbial species (Harmon et al. 1986). Furthermore, the nutrients and energy released during the decomposition of wood contributes to the maintenance of long-term site productivity.

At the time of a regeneration harvest (i.e., stand initiation), down and dead coarse wood in a forest stand comes from one of two sources. Dead coarse wood mass present at the time of stand initiation is residual dead wood from the previous stand or is created by the regeneration harvest itself (Janisch and Harmon 2002). Immediately following creation of early successional habitats, or during stand initiation (Fig. 15.3, stand age = 0), carbon stored in dead coarse wood is typically abundant due to woody residues left on-site during harvest operations (Spetich et al. 1999, Duvall et al. 1999). For example, in Central Appalachian hardwood stands regenerated via clearcutting, dead coarse wood can approach 55 Mg ha^{-1} 2 years post-harvest (McCarthy and Bailey 1994), which exceeds dead coarse wood biomass in many old-growth forests.

As aboveground biomass begins to accumulate in recently regenerated stands, low mortality rates of the arborescent vegetation layer, coupled with decomposition of residual and post-harvest dead coarse wood, reduces carbon stored in dead coarse wood to a minimum (Fig. 15.3, I). Aggregation of coarse wood begins slowly during stem exclusion (Oliver and Larson 1996); however, inputs are relatively low, given mortality during this stage of stand development is limited to intermediate and suppressed (i.e., small diameter) trees (Fig. 15.3, II). During understory reinitiation

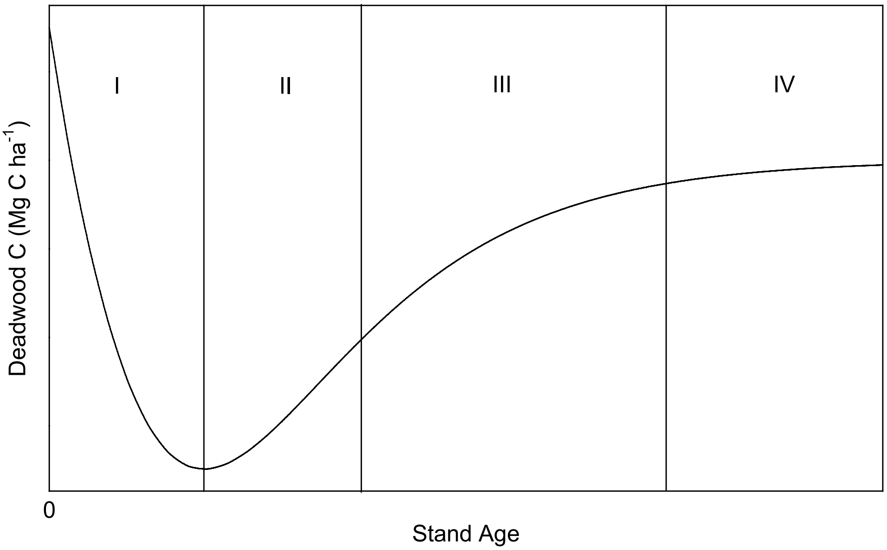

Fig. 15.3 Generic pattern of dead wood biomass and carbon (C) accumulation following a regeneration harvest as a function of stand age

(Oliver and Larson 1996), mortality of large-diameter trees beings to occur and the rate at which coarse wood carbon accumulates increases (Fig. 15.3, III) until an equilibrium between production and decomposition occurs during late (i.e., old-growth) stages of stand development (Fig. 15.3, IV). So, although regeneration harvests generally result in a short-term pulse of coarse wood to the system via logging residues, over time there is a reduction of both biomass and carbon that is sustained until stands enter the stand reinitiation or old-growth stages of stand development (Hardt and Swank 1997; Spetich et al. 1999; Duvall and Grigal 1999).

Although the U-shaped pattern (Fig. 15.2) of coarse wood accumulation following a regeneration harvest can be used to generalize coarse wood carbon dynamics across geographic regions and forest types (Harmon et al. 1986), the rate and amount of coarse wood carbon accumulation within any given forest stand varies as a function of time since disturbance or stand age (Sturtevant et al. 1997), type and severity/intensity of disturbance (Spies et al. 1988; McCarthy and Bailey 1994; Duvall and Grigal 1999), site quality (Harmon et al. 1986; Spetich et al. 1999), and decay rates, which are influenced by species, size of substrate, topographic position, climate, and site quality (MacMillan 1981; Harmon et al. 1986,1995; Janisch et al. 2005; Webster and Jenkins 2005; McCarthy et al. 2001; Spetich et al. 1999). The high degree of spatial variability in the coarse wood carbon pool is demonstrated by estimates of coarse wood volume (substitute for biomass) in old-growth forests that range from 32 m^3 ha^{-1} in Midwestern oak-hickory forests (Spetich et al. 1999) to 53 m^3 ha^{-1} in highly productive cove hardwood forests of Great Smoky Mountains National Park (Webster and Jenkins 2005).

15.2.4 Forest Floor

The forest floor comprises the upper portions of the soil profile defined by the litter (Oi) and duff (Oe + Oa) layers as well as small (<7.5 cm in diameter) dead woody biomass (IPCC 2006). Although vital to ecosystem function and nutrient cycling (Vitousek and Sanford 1986), the forest floor constitutes a minor component of a forest stand's overall carbon budget (Bolstad and Vose 2005; Li et al. 2007; Bradford et al. 2009). The distribution and abundance of the forest floor, as well as the response of this carbon pool to disturbance, is spatially and temporally variable (Wallace and Freedman 1986). Earlier models describing the response of the forest floor to disturbances suggested that as much as 50% of the forest floor mass is lost within the first 20 years, with recovery to pre-disturbance levels not occurring until approximately 50 years post-harvest (Covington 1981). Although this predicted pattern of forest floor mass and carbon loss and recovery has been observed elsewhere in eastern USA forests (Federer 1984; Mattson and Smith 1993; Griffiths and Swanson 2001), many studies in the Central Hardwood Region have documented long-term increases or no effect of disturbance on forest floor mass and, consequently, carbon storage, following regeneration harvests (Mattson and Swank 1989, Knoepp and Swank 1998, Johnson et al. 1991; Elliott and Knoepp 2005; Li et al. 2007). Increases in forest floor mass observed in forests across the Central Hardwood Region (Johnson et al. 1985; Mattson and Swank 1989; Johnson and Todd 1998, Knoepp and Swank 1998) following silvicultural treatments are attributed to logging debris deposited on the forest floor during harvest operations, increased fine-root mortality and production, and changes in species composition that may occur during stand development (Yin et al. 1989a, b; Knoepp and Swank 1997; Idol et al. 2000; Jandl et al. 2007; Nave et al. 2010).

Clearly, post-disturbance dynamics of the forest floor are variable. However, following a regeneration harvest, and provided successful regeneration occurs, a general pattern in which forest floor mass and, consequently carbon stored in the forest floor pool, increases until equilibrium between litter input and decomposition is achieved during the later stages of stand development. The recovery of forest floor mass following a stand initiating disturbance is a function of the quantity and quality of litter, climatic controls on decomposition (Prescott et al. 2000), and the relative speed at which aboveground biomass and leaf area of a regenerated stand develop (Bradford et al. 2009). Inputs of forest floor material can approach pre-harvest levels within a few years of a regeneration treatment (Covington and Aber 1980; Boring et al. 1988; Boring and Swank 1986), but this timeframe varies with site productivity (Vose and Allen 1988; Shi and Cao 1997; Frazer et al. 2000; Jokela and Martin 2000). Across the Central Hardwood Region, forest floor mass and the corresponding carbon pool are highly variable due to differences in decomposition rates and/or species composition and associated litter quality. In productive upland forests of the Southern Appalachians, forest floor mass has been found to range from 22.2 Mg ha^{-1} in 20 year old stands to 26.0 Mg ha^{-1} in an 85 year old stand to 26.7 Mg ha^{-1} in an old-growth stand (Vose and Bolstad 2007). In comparison, forest floor mass in less productive forest types is considerably less, ranging from only 13.1 Mg ha^{-1} in mature oak-hickory forests of Illinois (Luvall and Weaver 1986) to 20.0 Mg ha^{-1} in mixed-oak forests in southeastern Ohio (Graham and McCarthy 2006).

Forest floor dynamics following traditional approaches to creating early successional habitats, such as clearcutting, depend on the immediate effects of the treatment on the forest floor layer. Forest floor mass can decrease a result of decreased litter inputs or increased decomposition; alternatively, forest floor material can increase or be buried or mixed with mineral soil as a result of harvest operations. If treatment results in an immediate decrease in forest floor mass (Covington 1981), mass and carbon storage of the forest floor will decrease until, at some point in stand development, litter inputs exceed rates of decomposition. If, however, decomposition rates decrease or remain unchanged and the addition of logging debris offsets any short-term losses in litter inputs (until the point in stand development when litter input recovers to pre-disturbance levels), little to no change in forest floor mass or carbon storage capacity of the forest floor relative to pre-disturbance conditions can be expected over the short- and long-term (Knoepp and Swank 1997; Hall et al. 2006; Boerner et al. 2008).

15.2.5 Soil Organic Matter

Mineral soil carbon is commonly cited as the forest carbon pool that stores the greatest proportion of ecosystem carbon (e.g., Turner et al. 1995; Dixon et al. 1994). Although studies from mixed-hardwood forests in the Southern Appalachian region confirm this statement (e.g., Bolstad and Vose 2005), other forest types within the Central Hardwood Region store significantly more carbon in aboveground vegetation than in the mineral soil (Li et al. 2007). This suggests the capacity of the mineral soil to sequester carbon is spatially variable across physiographic regions. Accumulation of carbon in mineral soil depends on inputs from litterfall and fine root turnover, while release of carbon depends on mineral soil carbon quality (e.g., labile or stable condition) and edaphoclimatic conditions (Jandl et al. 2007). Despite the spatial (e.g., Sun et al. 2004) and temporal (e.g., Knoepp and Swank 1997) variability in size and significance of the mineral soil carbon pool across the Central Hardwood Region, independent studies (Mattson and Swank 1989; Knoepp and Swank 1997; Johnson et al. 2002a; Gilliam et al. 2004; Li et al. 2007; Boerner et al. 2008) and large-scale meta-analyses (Johnson and Curtis 2001) suggest that with successful vegetative recovery and accumulation of aboveground biomass, the mineral soil carbon pool is stable relative to pre-treatment conditions under a wide variety of silvicultural treatments, including regeneration treatments that create early successional habitats.

15.3 Conclusions

At the forest- or landscape-level, carbon storage is maximized when all stands comprising a forest are in an old-growth state. This, however, rarely occurs as natural or silvicultural disturbances create and maintain a heterogeneous mix of stands of various age classes and stages of stand development on the landscape. On a stand-by-stand basis, silvicultural disturbances can have a negative, albeit, short-term, impact on forest carbon storage relative to pre-disturbance levels, with the removal of aboveground biomass responsible for the greatest proportion of carbon lost. Forest management activities that may compensate for a proportion of the stand-level loss in carbon stocks include prescriptions that maintain partial canopy cover. Silvicultural methods that create early successional habitats while retaining structural heterogeneity in the post-harvest stand (e.g., Franklin 1989) may ameliorate some short-term losses in the aboveground carbon pool as well as alleviate some longer-term losses in the dead wood biomass pool that occur following clearcutting. Structural diversity can be increased by implementing a 2 aged silvicultural system or shelterwood with reserves (Smith et al. 1997), in which species of wildlife value (e.g., oaks in the Central Hardwood Region; see Greenberg et al., Chap. 8) are left in the overstory at densities low enough to regenerate a new cohort. As the newly regenerated cohort develops, the residual overstory or reserve trees provide structural diversity and wildlife habitat, as well as a source of large-diameter coarse woody debris, as these trees senesce and die throughout the rotation.

Although regeneration harvests used to create early successional habitats can convert stands from a carbon sink to a carbon source in the short-term, with successful vegetative recovery and adequate time (at least one full rotation) without disturbance(s), carbon storage will recover to pre-harvest levels. The timeframe for recovery of the various carbon pools will vary among forest types, edaphoclimatic conditions (Campbell et al. 2004), and the proportion of aboveground biomass removed (e.g., clearcutting versus single-tree selection). The timeframe during which an individual stand remains a carbon source will depend upon the amount of time required for carbon sequestration rates in the aboveground biomass pool to surpass carbon lost (Fig. 15.1) via decomposition of belowground, dead wood, forest floor, and/or mineral soil carbon pools.

In a landscape as diverse as the Central Hardwood Region, stand-level responses to disturbance, including responses that affect carbon storage, vary within and among physiographic regions. In highly productive forests, the amount of time during which a regenerated forest stand remains a net source of atmospheric carbon (i.e., negative net ecosystem productivity; Fig.15.1) may be minor due to the rapid accumulation of aboveground biomass and carbon (Davis et al. 2009). Less favorable edaphoclimatic conditions coupled with slower-growing species may result in a regenerated stand being a carbon source for a substantial period of time (Campbell et al. 2004). Regenerating high quality sites, however, has the potential to store more of the removed merchantable carbon in long-term products compared to less productive forest stands where timber quality and merchantability are lower (e.g., Gonzalez-Benecke et al. 2010). Because harvested wood products can constitute a large and long-term carbon sink (Skog 2008), failure to recognize and account for the carbon stored in durable, long-lived products would ultimately underestimate total carbon storage following forest management activities and overestimate carbon flux to the atmosphere (Nunery and Keeton 2010). When carbon storage in products is taken into consideration, the difference in carbon storage achieved and often maximized under a no management alternative versus that obtained through more active management is reduced (Johnsen et al. 2001; Seidl et al. 2007; Nunery and Keeton 2010).

When examining changes in carbon storage in relation to forest management, it is important to assess the carbon consequences of a single, stand-level management action at the landscape level (Harmon 2001). The effects of management on landscape-level carbon storage depend on the sum of stand-level changes in carbon stocks. Across a forested landscape, management actions that result in a short-term loss of carbon in any given stand are often offset by carbon gains in other, undisturbed stands (Harmon 2001; Ryan et al. 2010). The age-class distribution of forest stands has a particularly large effect on landscape-level carbon storage. Therefore, the impact of creating early successional habitats on forest- or landscape-level carbon storage will depend upon the temporal distribution and spatial scale of stands in young age classes on the landscape. A significant increase in the proportion of the forest or landscape in young age classes may negatively affect carbon storage in the short-term (e.g., Campbell et al. 2004; Depro et al. 2008). However, given stand-level reductions in carbon storage are largely attributed to reductions in aboveground biomass, upon successful regeneration and a period of re-growth equal to that of a full rotation length, stand-level carbon storage will approximate pre-disturbance levels. Although maximizing forest carbon is accomplished through eliminating disturbance or more passive management, the consequences of managing solely for carbon storage across a landscape may negatively impact other ecosystem services (e.g., Ryan et al. 2010), including creation and maintenance of wildlife habitat.

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