Managing Carbon Sequestration and Storage in Temperate and Boreal Forests

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Executive Summary

If carbon stocks and fluxes in temperate and boreal forests are to be included among efforts to mitigate global climate change, forest managers and policy makers must understand how management affects the carbon budgets in these systems. This chapter examines the effects of management of carbon sequestration, storage, and flux in temperate and boreal forests.

Existing Evidence Reveals the Following Trends

- Drainage of wetlands for increased tree production can result in either net carbon gain or loss, depending on how deep the drainage.
- Silvicultural thinning causes a reduction of the vegetative carbon pool, which recovers over a matter of decades, while the impact on soil carbon is considered limited.
- In certain forest systems, fuels reduction treatments (such as thinning and prescribed fire) result in lowered vegetative carbon storage, but result in forest structures that are significantly less susceptible to stand-replacing disturbance and the commensurate carbon releases from disturbance.
- Regeneration harvests significantly reduce the carbon stocks in vegetation and cause a transient increase in soil respiration, although the annual rate of carbon uptake will be greater in the regenerating stand. Harvested areas often remain net carbon sources for 10–30 years, then return to sinks.
- Carbon sequestration can be increased by extending rotation lengths, especially if maximum biomass productivity has not yet been reached.
- Fertilization can increase carbon storage in vegetation and reduce soil respiration rates, however gains are offset by the carbon released during fertilizer production.

We identified the following key points to consider for carbon storage and sequestration projects in temperate and boreal forests:

- Many forest management activities result in net carbon release and thus cannot demonstrate carbon additionality. Mechanisms should be developed to credit projects that reduce carbon loss, in addition to those that increase carbon gain.
- Where baselines are set for forest carbon project accounting determines which management activities are incentivized.
- The risk of carbon leakage must be addressed. If sequestration strategies simply displace timber harvests from one forest to another, at any geographic scale, carbon gains are neutralized.
- The amount of carbon stored in forest products, emissions from management operations,

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and fossil fuel displacement by forest biomass determine whether or not practices like thinning are positive, neutral or negative.

- Many forest management practices have minimal impacts on the soil carbon pool, which is the most difficult pool to measure. Thus, it may be possible that projects involving certain practices could avoid strict quantification of this pool.

1 Introduction

Forests play a major role in the mitigation of climate change, primarily through their ability to assimilate carbon dioxide and sequester it in living tissue, and in their long-term contribution to soil carbon stocks. Temperate and boreal forests are also a significant source of carbon emissions because of wildlife (Wiedinmyer and Neff 2007) and other disturbances (e.g., Zeng et al. 2009). Forest systems cover more than 4.1 billion hectares – approximately one third of the earth's land area (Dale et al. 2001) – and temperate and boreal forests make up roughly 49% of this total. Forests account for 90% of all vegetative carbon in terrestrial ecosystems and assimilate 67% of the total CO_2 absorbed from the atmosphere by all terrestrial ecosystems (Gower 2003).

Whether forests are sinks or sources of terrestrial carbon depends on the balance of processes that cause carbon sequestration (i.e. photosynthesis, peat formation) and release (i.e. increased respiration, forest disturbance). Taken as a whole, the temperate and boreal forest biomes were carbon sinks during the 1980s and 1990s (Schimel et al. 2001), but this may no longer be the case because the Canadian lodgepole pine forests are poised to release massive amounts of carbon as the result of die-off from insect infestations (Kurz et al. 2008). The moist temperate forest sink has been consistently growing with the abandonment of marginal agricultural lands (Houghton et al. 2000), and does not experience the same scale of disturbance-mediated carbon release as in the boreal or inter-mountain forests.

The emphasis on silvicultural practices in boreal and temperate forests is appropriate because increasing forest carbon stocks in these regions is a matter of making adjustments to existing forests and not undergoing radical changes in land use. Most temperate forests are second growth (Whitney 1996), much of the boreal has recently been cutover, but land conversion is minimal when compared to other regions of the world. Therefore, providing additional carbon storage is a matter of refining silvicultural practices, better quantifying the effects of disturbances, and examining the storage potential of forest products.

2 Boreal and Temperate Forests of the World

Boreal forests comprise the northernmost forest biome of the world, covering much of Alaska, Canada, Fennoscandia, Russia, northern Mongolia and northeast China. Boreal forests are characterized by simple, often single layered stand structure, low tree species diversity (only six genera dominate the entire range: spruce (*Picea*), fir (*Abies*), pine (*Pinus*), larch (*Larix*), birch (*Betula*) and aspen (*Populus*)) and well-developed bryophyte (moss and lichen) communities. Organic-rich peat soils in boreal forests and bogs (histosols or spodosols) are the largest carbon pool in the biome.

Boreal forests can be roughly divided into two major zones – interior continental and maritime (Fig. 10.1). As the name implies, interior continental forests are exposed to cold, dry continental climates. Fire and large-scale insect outbreaks are the dominant disturbance agents. In North America, interior continental boreal forests are dominated by white spruce (*Picea glauca*), Jack pine (*Pinus banksiana*), and aspen (*Populus tremuloides*) in different mixtures. In Eurasia, interior continental forests are found east of the Ural Mountains. Siberian larch (*Larix sibirica*) and Dahurian larch (*Larix gmelinii)*, both adapted to extreme cold, drought, and permafrost, cover much of this area.

Maritime boreal forests are found in North America along the Pacific coast (Cordillarean type) and Atlantic coast (Maritime type). In this moderated climate, fir species compose a larger proportion of forest area, and fire gives way to insect outbreaks and commercial harvesting as the primary disturbance agents. Maritime forests are also found in Fennoscandia and northwest Russia near the Norwegian, Baltic and White Seas. Scots pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*)

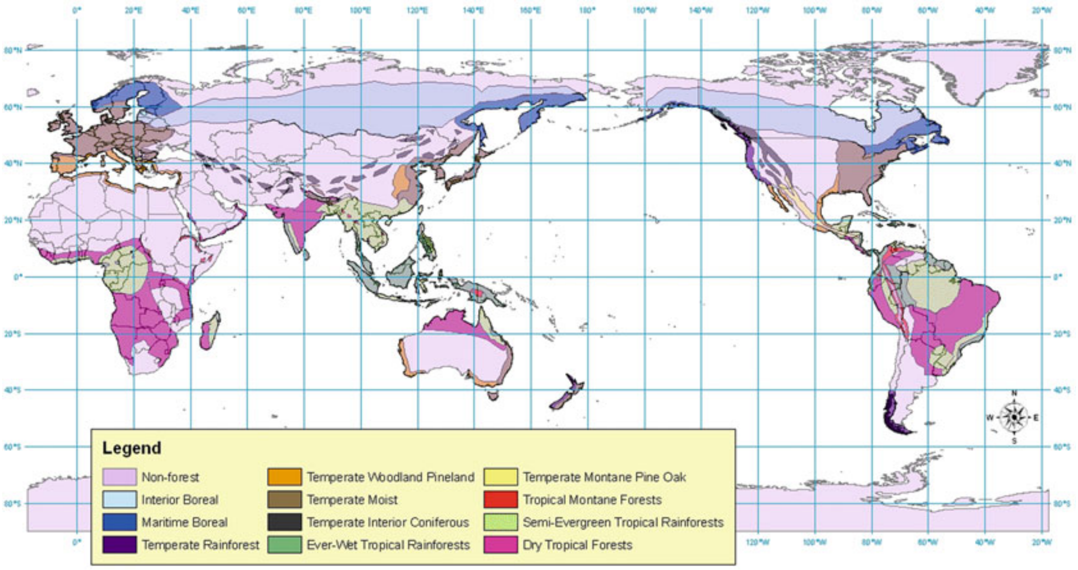


Fig. 10.1 Original extent of boreal, temperate, and tropical forest types of the world prior to land clearing

are the canopy dominants, with a considerable component of aspen and birch. Ground fires, insect outbreaks, and harvesting are major influences.

Temperate forests include a wide range of forest types across the midlatitudes, and the boundaries with boreal forests to the north and tropical forests to the south are subject to interpretation. With a distinct but relatively mild winter, temperate forests are characterized by more diverse climatic conditions and angiosperm species than in the boreal forest type. Generally speaking, the soil carbon pool does not play as large a role here, while the prominence of the vegetative pools increases.

There are five major temperate forest types:

1. *Moist broadleaf and coniferous forests*: mesic, mixed forests with a rich suite of genera, including maple (*Acer*), Oak (*Quercus*), birch (*Betula*), beech (*Fagus*), ash (*Fraxinus*), poplar, aspen (*Populus*), hemlock (*Tsuga),* "soft pines" (*Pinus*), spruce (*Picea*) and fir (*Abies*). Fire plays a relatively minor role in such forests. They are located in the eastern United States and Canada, northern and central Europe, and the Russian Far East. Soils classified as ultisols (USDA 1975) underlie much of this area, particularly in North America, and are generally desirable for cultivation because they are usually relatively fertile (though often stony) and require no irrigation because of precipitation year round.
2. *Interior coniferous forests*: dry, fire-adapted forests in harsh continental mountainous climates, with soils that are inceptisols (glacial non-volcanic) or often andisols (volcanic). "Hard pines" (*Pinus*), spruce, fir and larch predominate. Located in the interior west of the USA and Canada, and in Central Asia, these forest types are closely related to interior continental boreal forests. Soils are young, rocky, often skeletal, and exposed to the extremes of cold winters and dry summers.
3. *Montane oak/pine forests*: *Pinus*- and *Quercus*-dominated systems in mountain ranges of Mexico and Central America, the Himalayas, the Mediterranean and Turkey. They are fire-adapted and relatively dry. Soils are mixed.
4. *Woodland and pineland forests*: Fire-adapted, often open forests in dry, southern climates. They include "hard" pine forests of the U.S. coastal plain, pine and oak in the coastal Mediterranean region, *Acacia-Eucalyptus* savannas of Africa and Australia, and oak woodlands. Soils that are generally classified as alfisols (USDA 1975) predominate. Such soils are more fertile than ultisols but often require partial irrigation because of drier summers. Most forests with alfisols have already been cleared for cultivation, thus this type is restricted to degraded relics.

5. *Temperate rainforests*: Mesic, constantly moist, and often extremely productive forests of mountain ranges along coasts. Spruce, hemlock, Douglas fir (*Pseudotsuga*) and western cedar (*Thuja*) dominate in the Pacific Northwest, the southern beech (*Nothofagus*) in Chile, and southern beech, Eucalypts (*Eucalyptus*) and podocarps (*Podocarpus*) in New Zealand and Australia. Spodosols and andisols are the predominant soil types. Andisols are volcanic soils that with high precipitation can be very productive for pasture. Spodosols are acidic soils associated with bedrock geology that predominantly comprise minerals such as quartz and silica, and are therefore often nutrient poor.

3 The Forest Carbon Cycle

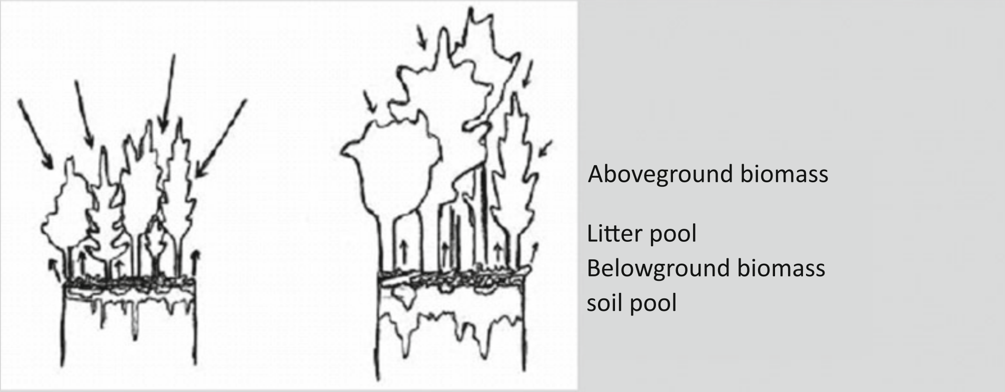
The following concepts pertain to the basic biological dynamics of carbon uptake, storage, and release, and also to important differences in how carbon pools in managed forests are quantified.

3.1 Maximizing Carbon Uptake vs. Maximizing Carbon Storage

Biomass productivity is maximized relatively early in forest development, at the time when annual growth increment dips below the average annual growth increment over the age of the tree or stand. After this point growth slows, and carbon uptake slows along with it. However, while older trees (and stands) may demonstrate reduced uptake rates, the carbon stored within them can greatly exceed that of their younger, perhaps faster-growing, counterparts. Greater pools of soil and litter carbon in older forests may also contribute to this effect, although their pattern is less clear than that of the vegetative pool.

The importance of this difference lies in its management consequences. Managing for productive

Insert 1. Maximizing C Uptake Versus Maximizing C Storage



These two images demonstrate the contrasting strategies of growing vigorous young forests with high rates of carbon uptake (left), and growing forests to older age classes at which uptake rate is lower, but actual quantities of stored carbon are greate (rights). The downward pointing arrows indicate carbon uptake through photosynthesis, the rates of which are indicated by arrow size. Upward arrows indicate C release through auto- and heterotrophic soil respiration. In the old forest shown on the right, the inputs and outputs are near equilibrium, while on the left, uptake clearly exceeds carbon loss. However, note that the actual size of the aboveground biomass, litter and belowground biomass are considerably larger in the older forest. Importantly, the size of the soil pool does not differ much between the two examples.

young forests promotes maximal carbon uptake, while maintaining old forests and extending rotations leads to larger on-the-ground carbon stocks. In theory, a series of short rotations can sometimes lead to greater total carbon storage than a single long rotation because the stand is growing at a rapid rate for a greater proportion of the time. But each harvest entry is also followed by a release of carbon associated with decomposition.

3.2 Site and Climatic Factors Limit the Carbon Storage Potential of Vegetation

In any given forested site, the maximum potential productivity and carbon storage of vegetation is determined by soil fertility, moisture conditions, and climate. These factors can be regarded as placing a "ceiling" on biomass production. Forest managers can manipulate and re-allocate biomass in different assemblages of species and stand structures. But to create additional carbon storage requires addressing the basic productivity constraints, for instance by fertilizing, irrigating, or draining the site.

A major caveat, however, is that forests may not reach their "biomass ceiling" for hundreds of years, often much longer than the rotations used in conventional forest management (Luyssaert et al. 2008). There are a number of forest management strategies that increase carbon storage (Evans and Perschel 2009). For example, it is often possible to gain carbon benefits simply by growing forests on longer rotations so that they have time to accumulate higher standing volumes (Foley et al. 2009).

3.3 The Carbon Impact of an Activity Changes if the Forest Products Carbon Pool is Included

Thinning results in a reduction of the vegetative carbon pool. It is possible that the residual trees will eventually replace the biomass lost in a harvest, and the pool will equal or exceed its pre-treatment storage. But due to the productivity constraints described above, the pool will never exceed the storage potential of the stand if it had never been thinned. This makes thinning a carbon-negative or at best carbon-neutral activity *unless* the sequestration of carbon within forest products is considered – that is, products are considered to be another "pool" (Eriksson et al. 2007). When the product pool is included, thinning can become carbon-positive because some portion of the harvested carbon will be stored in long-term forest products, while the residual trees are growing at a faster rate and taking up more carbon (e.g. Finkral and Evans 2008).

The inclusion and quantification of the forest products pool in carbon offset programs are topics of much debate and discussion. It is important to recognize the impact that this pool can have on the measurement of the carbon in forest management practices. A comprehensive discussion of the forest products pool is provided in Chapter 12.

3.4 Resiliency: Maximum Carbon Storage at High Risk vs. Reduced Carbon Stocks at Reduced Risk

Forest managers have long recognized that maximizing the density of biomass on a site can be detrimental to forest health. Density-related competition often results in spindly, poorly-formed trees that are not windfirm, are susceptible to insect outbreak, and pose fire risks. On a larger scale, the risk of such disturbances is also increased when a large proportion of the landscape is maintained in dense stands within a limited age class range. Foresters address these concerns by managing for stand- and landscape-level resiliency. Stands are often managed at lower than maximum densities, in order to reduce risk of catastrophic loss. A sacrifice in biomass production is made in order to produce fewer, larger, more vigorous trees.

This principle still applies when carbon uptake and storage is the management goal. Carbon stored in fire-, insect- or windthrow-prone trees and stands is "risky," and some sacrifice in total storage may be necessary to ensure that sequestration is long-term.

3.5 Creating Carbon Additionality vs. Minimizing Carbon Loss

Because of the structure of many carbon offset programs, the primary goal of managing forest carbon is often to create additionality. Certain practices are regarded as reliably "additional," such as afforestation (unless by changing the site a large soil carbon loss is incurred). However, the manipulation of standing forests more commonly results in immediate reductions of carbon pools. Such practices can be adapted in certain ways to reduce their negative carbon impact, such as by leaving more harvest residues or causing less damage to residual trees during harvest. This can result in a form of additionality, compared to business-as-usual management techniques. Activities such as reduced deforestation and reduced impact logging appear additional when compared to such a business-as-usual baseline.

4 Carbon Impacts of Specific Forest Management Practices

4.1 Application of Resiliency

Disturbance plays a vital role in the natural flow of carbon between pools, but as a result of past management practices and a changing climate, many forests in the boreal and temperate regions have become especially susceptible to catastrophic disturbances (Hurteau and North 2009) that release large pulses of carbon into the atmosphere.

Managing for carbon should strive to maximize the amount of stored carbon while minimizing the likelihood of stand-replacing disturbance. This balance is achieved through maximizing forest resiliency, the capacity of a system to absorb disturbance and reorganize while undergoing change so as to retain essentially the same function, structure, and ecosystem services (Folke et al. 2004). This definition works well for carbon purposes because it accounts for a resilient forest's ability to reduce carbon loss from a disturbance and reorganize in such a way that maintains high levels of the desired ecosystem service, carbon sequestration. Here are examples of management responses to four very common disturbances in boreal and temperate forests: fire, wind, insect infestations and climate change.

4.1.1 Fire

Fire is a dominant disturbance agent in many temperate forest regions. In some regions, uncharacteristic fire frequency and intensity is due to changing climactic conditions (Lucas et al. 2007). In many others, the structure of fire dependant temperate forest ecosystems has been altered as a result of a high level of fire suppression over the last 100 years (Covington et al. 1997; Allen et al. 2002; Brown et al. 2004). This has resulted in a buildup of fuels leading to intense fires (Hessburg et al. 2005). Tilman et al. (2000) found that in an oak savannah in Minnesota, when fire was excluded, forests were able to build both above and belowground biomass to levels 90% greater than in forests with frequent ground fires. This sequestered carbon is at high risk of sudden release due to the potential for stand-replacing fire. On such sites, forest managers may choose to balance increased sequestration with increased stability by reducing stem density and fuel loading.

The restoration of more fire-resilient forests is possible and critical (Agee and Skinner 2005). A combination of thinning and burning can build resiliency through the removal of accumulations of biomass fuels at sites. Forests under such management will store less carbon than the maximum possible, but over the long term they may store more than forests that experience stand-replacing fires (Houghton et al. 2000). In the southwestern U.S., a thinning designed to reduce fire risk reduced the total amount of carbon stored in a ponderosa pine stand and turned it into a weak carbon source for a short period following treatment (Dore et al. 2010). Although the carbon sink strength was reduced, the reductions in total stand carbon and gross primary productivity were not as much as in a nearby stand that experienced a high-intensity fire. Furthermore, the thinned stand can continue greater levels of primary production compared to the burned stand (Dore et al. 2010). It is well known that fire severity determines the amount of carbon released during the acute stages of the disturbance. However, some studies indicate that nearly half of the carbon released is lost through the much slower decomposition processes over a period of years (Brown et al. 2004; Hessburg et al. 2005). In fact, some experiments have shown that recently burned and harvested sites are sources of carbon, and that recovery to the same flux as a mature site can take 10 years following a fire (Amiro 2001). Causes of this phenomenon are linked to an increase in soil respiration due to an increase in soil surface temperatures. The complex interactions between fire, soils, vegetation, and site recovery from a disturbance are just beginning to be understood.

Prescribed fire treatments are intended to reduce fuel loads without causing significant mortality to the remaining vegetation. It is important to point out that there is a carbon loss associated with the use of prescribed fire. Surface soils, litter and downed woody material will be carbon sources for some years after the disturbance. Land managers need to weigh these emissions against either a no-action alternative or another silvicultural treatment to determine the best fit for the site. It should be stressed that the carbon loss from a high-intensity fire can be extensive and long-lasting.

Some boreal and temperate forest types, such as lodgepole pine (*Pinus contorta*), have evolved with stand-replacing wildfire. It would thus be misguided to attempt to produce more resilient forest structures – ones “capable of maintaining substantial live basal area after being burned by a wildfire” (Agee and Skinner 2005) – in fire-dependent ecosystems. The autecology of species like *P. contorta* may make stands they dominate inherently more “risky” for carbon sequestration, and inappropriate as sites for long-term storage.

4.1.2 Wind

Unlike fire, the magnitude of carbon loss from a wind disturbance is not so closely linked to stocking density. Wind as a disturbance agent can affect forests through a wide range of magnitude and spatial scales, from a localized downburst damaging a single tree to the large-scale damage caused by hurricanes (McNulty 2002). Over the period 1851–2000 tropical cyclones caused an average carbon release of 25 Tg/y (Zeng et al. 2009). The resilience of trees and understory vegetation to wind disturbance can provide a tight biotic control of ecosystem processes like carbon sequestration, and is based on the structure of the forest prior to the disturbance (Cooper-Ellis et al. 1999). The greater the diversity of functional groups represented in the pre-disturbance forests, the greater capacity the forest has to maintain or recover the ability to sequester carbon in the environment that follows the disturbance (Busing et al. 2009).

4.1.3 Insects/Pathogens

In recent decades there has been no shortage of examples of both native and exotic pests and pathogens causing tree mortality in boreal and temperate forests. Exotic pests and pathogens have great potential to alter forest carbon dynamics (Peltzer et al. 2010; Ayres and Lombardero 2000). Depending upon species-specific characteristics, mixed forests may contribute to ecological stability by increasing resistance and resilience (Larsen 1995). A good example is the mixed hemlock/hardwood forests of the northeastern USA. Hemlock woolly adelgid attacks hemlock trees of all ages and sizes, and infested trees seldom recover (Nuckolls et al. 2008). Carbon effects from the infestation are not surprising; during the first year of infestation, autogenic respiration of CO_2 from roots is reduced although no additional carbon is stored because there is little or no photosynthesis occurring. Decomposition increases as trees die as a result of increased light regimes, leading to increased soil temperatures. Overall the carbon release depends on the size of the infestation and the species mix associated with the hemlock stands. Since most hemlock stands are not single species, or single age class the carbon loss from the ecosystem as a whole is less than in monotypic forest types such as lodgepole pine (Albani et al. 2010; Orwig and Foster 1998). Additionally, large-scale stand-replacing fires are not typical in the eastern US where the hemlock woolly adelgid is found. In the context of carbon sequestration, mixed hemlock/hardwood forests are more resilient to insect infestation than lodgepole pine forests because of their diversity (Schafer et al. 2010).

Insert 2. Managing for Resiliency in Forests Affected by the Mountain Pine beetle

Managing for resiliency in forests affected by the mountain pine beetle

"There are literally several hundred million cubic meters of wood out there in the forests decomposing and releasing carbon dioxide back into the atmosphere," (Kurz et al., 2008) from a massive outbreak of the mountain pine beetle (Dendroctonus ponderosae) across the lodgepole pine (Pinus contorta) forests of interior British Columbia. This infestation and subsequent catastrophic fires in beetle-killed timber are threatening to turn Canada's forests from a carbon sink to a source. It is projected that the region could release 990 million tons to CO_2 *– more than the entire annual emissions reported by Canada in 2005 (Kurz et al., 2008).*

Research has demonstrated that direct management of mountain pine beetle through tree removal, burning or insecticide application is impractical and ineffective. Rather, that alteration of stand structure (age-class distribution, composition and density) has the best chance of minimizing the scale and intensity of the infestations and associated negative carbon flux from these forests (Amman and Logan, 1998). Unfortunately, because of a century long campaign of aggressive fire suppression, and an attempt to maintain a status quo of current stand conditions that goes beyond the natural cycle of regeneration and renewal, there are limited opportunities for appropriate silvicultural treatments.

4.1.4 Climate Change

If climate change alters the distribution, extent, frequency, or intensity of any of these disturbances, large impacts could be expected (Dale et al. 2001). For example, as climate changes, the ability of native and non-native forest pests to establish and spread increases because the range of suitable environment expands. The door opens to insects and pathogens that previously posed less of a risk. Direct effects of climate change on forest pests will likely be increased survival rates due to warmer winter temperatures, and increased developmental rates due to warmer summer temperatures (Hunt et al. 2006). A striking example is in the interior of British Columbia where the mountain pine beetle (*Dendroctonus ponderosae*) infestation is rapidly spreading to the north (Ayres and Lombardero 2000; Peltzer et al. 2010).

The diversity of species in an ecosystem undergoing change appears to be critical for resilience and the generation of ecosystem services (Folke et al. 2004). In this sense, biological diversity provides insurance, flexibility, and a spreading of risk (Duffy 2009). Therefore management should attempt to strive for diverse, mixed species, multiple age class stands, or any combination thereof, for all forest types – simple or complex. It is one important tool that contributes to sustaining the response required for renewing and reorganizing desired ecosystem states after disturbance (Larsen 1995).

Resilience can be influenced at the landscape level by the presence of refugia that escape disturbance and serve an important re-colonization function for surrounding areas. This diversity of species and heterogeneity in the landscape builds integrity, meaning that even if the disturbance causes a change in the stable state of the forest, the new stable state will function in a similar way, providing the same ecosystem services, including carbon sequestration (Perry and Amaranthus 1997).

4.2 The Concept and Application of Thinning

Thinning is a silvicultural practice that lowers stand density through the removal of a portion of the standing volume, often at regular spacing. Thinning clearly impacts the aboveground vegetative carbon pool, and it also affects the litter pool (through the addition of slash and reduction of post-thinning litterfall), and potentially the soil pool (through increased respiration due to increased light and warmth at the soil layer).

Thinning increases the amount of available growing space for residual trees, thereby leaving potential growing space vacant for a period of time immediately post-treatment, resulting in reduced stand carbon storage (e.g., Campbell et al. 2009; Spring et al. 2005; Nilsen and Stand 2008; Balboa-Murias et al. 2006). Importantly, the decrease in stand production does not always scale perfectly with the reduction in stand density. Light-use efficiency of ponderosa pine was almost 60% higher in thinned than unthinned stands (Campbell et al. 2009), perhaps because the trees removed in the treatment were of low vigor and were not using site resources efficiently. Also, if canopy thinning stimulates increased growth in midstory and understory vegetation, reductions in aboveground net primary production can be quickly offset (e.g. thinning in Ohio oak-maple (*Quercus-Acer*) stands, Chiang et al. 2008). However, after thinning, a stimulated shrub layer can also result in net carbon loss if it has lower net primary productivity than the tree layer but similar respiration rates (Campbell et al. 2009).

Different types and intensities of thinning have different impacts on carbon storage. For example, in Allegheny hardwoods, plots thinned from below showed no significant difference in carbon storage from unthinned plots, crown-thinned plots sequestered significantly less carbon, and thinned-from-above plots even less (Hoover and Stout 2007). A pre-commercial thinning in New South Wales increased total stand carbon because all the cut trees remained on the ground (and were sequestered for some time in the litter pool) while the residuals accumulated biomass at a faster rate (McHenry et al. 2006).

Thinning influences litter and soil carbon as well. In general, forest floor carbon declined with increasing thinning intensity in field studies in New Zealand, Denmark, and the USA (Jandl et al. 2007). Litterfall additions to the forest floor and higher ground temperatures stimulated decomposition. However, the impact was moderated by the addition of logging slash to the litter layer, and the fairly rapid return to pre-treatment temperatures in all but the most intensively-thinned plots (Jandl et al. 2007). Increases in CO_2 efflux after thinning have been observed for several years in California

Insert 3 Thinning and the C Balance of a Forest Stand

Thinning and the carbon balance of a forest stand

Flux tower measurements taken in a 40-year-old Scots pine (Pinus sylvestris) stand in southern Finland showed that CO_2 *flux did not change after the first commercial thinning. A complex of factors allowed this. A reduction in overstory photosynthesis was balanced by an increase in understory photosynthesis. And while heterotrophic respiration increased with the decomposition of logging slash and roots, this in turn was balanced by a reduction in autotrophic root respiration.*

Thus, the "redistribution of sources and sinks is comprehensively able to compensate for the lower foliage area" in the thinned stand.

From Suni et al., 2003

mixed conifers and Ozark oak-hickory (*Quercus-Carya*) stands (Concilio et al. 2005).

The soil pool appears even more buffered from the effects of thinning than the litter pool. Some increase in soil respiration was observed after thinning in Norway spruce, but no significant effects on soil carbon storage could be detected with increasing thinning intensity (Nilsen and Stand 2008). Thinning in South Korean *Pinus densiflora* and German European beech (*Fagus sylvaticus*) forests produced no significant increases in respiration (Dannenmann et al. 2007; Kim et al. 2009). In loblolly pine (*Pinus taeda*) plantations in Virginia, the contribution of logging slash and decaying roots to the soil actually *increased* soil carbon concentration in the 10–40 cm depth 14 years after thinning (Selig 2008).

Thinning thus produces a short term decrease in vegetative and litter carbon pools, and little to no increase in soil respiration. How long this negative impact on carbon storage on-site lasts depends on the intensity and type of thinning, and on how fast residual trees can replace the biomass removed. Whether slash inputs to the litter layer exceed reductions in litterfall also plays a small part in defining when pre-treatment carbon levels are re-attained.

4.3 Site Treatments

4.3.1 Drainage

Drainage is implemented where excessive soil moisture stunts or prohibits the growth of trees. Within the boreal and temperate zones, this practice is most prominent in Fennoscandia, particularly in Finland. Drained peatland forests constitute 18–22% of the total managed area of that country (Minkkinen et al. 2001). Afforestation of drained peatlands has also occurred on a large scale in Great Britain and the coastal mires of the southern United States. These peatland areas are associated with high levels of soil carbon storage, but also with emissions of CH_4 (methane), an important greenhouse gas.

The carbon consequences of land drainage depend on whether the factors that increase sequestration (increased vegetative production, increased litter input, and decreased methane release) exceed the increased respiration caused by oxidation of previously anoxic peat. A critical factor in this balance appears to be how much the water table is lowered in the drainage process. When the water table was lowered from 0–10 cm to 40–60 cm (below the surface) in Finnish mires, CO_2 loss increased 2–3 times and stayed at that rate for at least 3 years (Silvola 1986; Silvola et al. 1996). At this rate, Silvola (1986) found that such mires would switch from a modest carbon sink to a strong carbon source. Similarly, deep drainage of peaty moorlands in Britain for Sitka spruce (*Picea sitchensis*) afforestation would result in sufficient drying such that all but the recalcitrant peat component would decompose resulting in net carbon emissions (Cannell et al. 1993).

In contrast, when the water table in a Finnish mire was only lowered 5–9 cm, emissions barely changed (Silvola et al. 1996). Similarly, afforestation of Irish moorlands did not result in deep drying or oxidation and increased CO_2 release was minimal (Byrne and Farrell 2005). Von Arnold et al. (2005) examined CO_2 and CH_4 efflux (which are usually negatively correlated) in undrained, lightly drained and well-drained (dry) peatlands in Sweden. They found that, from the perspective of minimizing greenhouse gas emissions, the optimal condition was lightly drained peat, because increases in CO_2 efflux were exceeded by the decease in CH_4 efflux. In contrast, both undrained and dry peats were carbon sources to the atmosphere. Importantly, this analysis did not consider the additional sequestration potential of enhanced tree growth and litter production.

When the biomass and litter pools are considered, even greater carbon gains have been recorded in Sweden, Finland and Russian Karelia (Laine and Vasander 1991; Minkinnen and Laine 1998; Sakovets and Germanova 1992). Drained, plowed and afforested peatlands in Scotland were a carbon source for only 4–8 years, at which point increased vegetative productivity switched them to sinks. This effect only increased as the forests matured (Hargreaves et al. 2003).

Thus, drainage of peatlands for increased forest productivity has the potential to be carbon positive or carbon negative, depending on how thorough the drainage is. Shallowly drained sites tend to sequester more carbon than undrained sites because increased tree growth and decreased methane emissions outweigh increased CO_2 emissions. The opposite is true on deeply drained sites.

4.3.2 Fertilization

Tree growth in temperate regions is typically nitrogen-limited. Therefore, nitrogen fertilization is a well-established treatment in this region to increase biomass production. This increased capacity to store carbon is well documented, but must be considered in light of the carbon emissions required to produce and apply the fertilization treatment.

Biomass production is the result of the energy produced by photosynthesis, minus the respiration requirements of the non-photosynthetic plant tissues. Higher fertility increases leaf area, nutrient concentration, and carbon assimilation rates and in turn, improves carbon availability and overall biomass production (Coyle and Coleman 2005). Nitrogen fertilization has been shown to increase biomass production as much as 16 Mg ha^{-1} over 100 years in some intensively managed pine forests in the southeastern United States (Markewitz et al. 2002). On some low fertility sites, nitrogen fertilization can make the difference between the site's being a carbon source or a carbon sink and can lessen the time it takes for a developing stand to go from a source to a sink. The degree of effect that fertilization has depends on the baseline fertility of the site (Maier and Kress 2000).

The fertility of a site can be approximated by determining the nitrogen-use efficiency, a measure of the amount of additional carbon assimilated as a result of the addition of a kg of nitrogen. Nitrogen-use efficiency for carbon sequestration in trees strongly depends on soil nitrogen status as measured by the carbon/nitrogen ratio. Excessive fertilization or appropriate fertilization plus the deposition of anthropogenically elevated levels of atmospheric nitrogen can cause deposition rates to exceed the capacity for nitrogen uptake, and nutrient imbalances can lead to forest decline due to nitrogen saturation (Bauer et al. 2004). The effect of nitrogen saturation is also seen in soils when the biotic component of soil is no longer able to uptake and stabilize the nitrogen in organic compounds. The excess nitrogen is leached out of the soil in the form of nitrates (Magnani et al. 2007).

It has been thought that fertilization decreases soil carbon stocks through an increase in decomposition. However, many recent studies have demostrated that fertilization may increase carbon stocks in the soil. Hagedorn et al. (2001) found that soil organic carbon (SOC) sequestration in fertilized plots was always higher than that in control plots. They and others conclude that fertilization of temperate and boreal forests has high potential to reduce both heterotrophic and autotrophic soil respiration (Pregitzer et al. 2008). Decomposition is slowed as a result of several factors: (i) decreased carbon allocation to mycorrhizae; (ii) direct suppression of soil enzymes responsible for litter degradation; (iii) decreased litter quality; and (iv) decreased growth rates of decomposers. The research highlighting the sequestration of SOC as a result of fertilization is relatively recent and the hypotheses about the mechanisms that drive it are primarily speculation. More research is needed to address this knowledge gap.

Similarly to nitrogen fertilization, temperature can influence soil carbon stocks in the temperate and boreal regions. Temperature can influence nutrient availability and therefore fertility. In the future, therefore, the effect of nitrogen fertilization on soil carbon storage may be offset by the opposite effect of climate change; small increases in temperature will increase the rates of decomposition and nitrogen cycling and the carbon stock of forests may decline due to accelerated decomposition of SOC (Makipaa et al. 1999). This is likely to be a gradual change, but will be most pronounced in the boreal regions where processes are typically more limited by temperature than in temperate regions.

Although nitrogen is limiting in many forests of the temperate and boreal regions, it is not the only fertilization treatment used. In nitrogen-rich sites such as drained peatlands in central Finland or poorly drained loam and clay soils of the upper coastal plain of Georgia, USA, treatments such as additional phosphorus, calcium, potassium or liming are needed to amend critical nutrient levels or pH (Hytönen 1998; Moorhead 1998). In northeastern Oregon and in central Washington where nitrogen is considered limiting, research has shown that the addition of nitrogen and sulfur to Douglas-fir stands produced significant growth response to the nitrogen + sulfur treatment, but not to the nitrogen-alone treatment (Garrison et al. 2000). Similarly, in loblolly pine stands in the coastal plains of Georgia, USA, phosphorus is needed to enhance uptake of nitrogen (Will et al. 2006). Finally, in northwestern Ontario, Canada, the best treatment in terms of total volume increment over that of the control was 151 kg nitrogen ha^{-1} plus 62 kg magnesium ha^{-1}, which produced about 16 m^3 ha^{-1} of extra wood over 10 years (Morrison and Foster 1995).

These examples illustrate the complexities often associated with the correct application of fertilization and amelioration treatments to increase carbon on forested sites. These treatments are site specific; a manager's mastery of the intricacies of the site is essential to increasing the carbon uptake on a site.

It is beyond the scope of this chapter to provide a comprehensive look at the trade-offs between an increase in carbon storage in temperate and boreal forests and the fossil fuel emissions that result from the acquisition, manufacture, transport, and application of fertilizers. Most results indicate that even on the sites where fertilization is most beneficial, the emissions of CO_2 outweigh the carbon sequestered as a result of increased biomass production and SOC stocks (Schlesinger 2000; Markewitz 2006). However, on nitrogen-poor sites, where appropriate, the encouragement of the establishment of nitrogen-fixing plants may be beneficial through natural or artificial seeding (Marshall 2000).

4.4 Concepts and Application of Regenerating Forests

4.4.1 Afforestation and Reforestation

Afforestation and reforestation are silvicultural treatments that typically demonstrate carbon additionality. For example, the average net flux of carbon attributable to land-use change and management in the temperate forests of North America and Europe decreased from a source of 0.06 PgC yr^{-1} during the 1980s to a sink of 0.02 PgC yr^{-1} during the 1990s (Houghton 2003). In the United States this carbon sink is overwhelmingly due to afforestation /reforestation rather than active management or site manipulation (Caspersen et al. 2000). Even though some studies suggest that as forests age the strength of the carbon sink is reduced (and may become a source under certain circumstances), the amount of carbon stored on a forested site is significantly more than any other ecosystem type (Vesterdal et al. 2007).

Land conversion to forests is typically driven by wood demand and not carbon sequestration and it is unlikely that this will change even as carbon markets develop (Eggers et al. 2008).The conversion of land to forests using passive, natural regeneration has been postulated as an option for carbon sequestration because of the low operating costs and potential for co-benefits such as habitat and water quality enhancement (Fensham and Guymer 2009). These co-benefits provide valuable ecosystem services, but proving that the intent of the project was strictly for carbon sequestration (additionality) is complicated. Rules for proving additionality are not well established and/or uniform across carbon offset programs, so landowners planning to invest in afforestation/reforestation for the purpose of capturing market benefits need to make clear that the intent of the project is to sequester carbon.

4.4.2 Regeneration Harvests

Regeneration harvests are silvicultural treatments that remove some or all of the existing forest overstory to release existing regeneration or make growing space available for the establishment of a new cohort. Regeneration harvests alter the aboveground vegetation, with the added potential of affecting the bryophyte, and litter carbon pools; and potentially the mineral soil carbon.

The effect on the vegetative pool depends on the type of regeneration harvest. Uneven-aged treatments such as selection harvesting may have effects similar to thinning in that they only remove a portion of the canopy cover (Laporte et al. 2003;

Harmon et al. 2009). In a comparison of harvest types in Ontario, Canada, carbon storage in northern hardwoods was greater after selection harvesting than clearcutting because vigorous residual trees remained on the site (Lee et al. 2002). Clearcutting has a distinct and stronger effect. A clearcut of old-growth Norway spruce in Finland resulted in a 1/3 reduction in ecosystem carbon (Finer et al. 2003). Whole-tree harvesting on a 100-year rotation was modeled to result in an 81% reduction in biomass carbon compared to uncut forests in boreal China (Jiang et al. 2002).

Harvesting's influence on litter and particularly mineral soil carbon is controversial. An influential study by Covington (1981) in clearcuts at Hubbard Brook Experimental Forest in New Hampshire showed increased decomposition (and hence soil carbon loss) after forest harvest, suggesting that forest floor organic matter declines 50% within 20 years of harvest. A number of studies reinforce this view. In a modeling simulation of the effects of different harvest regimes on carbon stocks in boreal *Larix gmelinii* forests in China, clearcutting was predicted to result in litter and soil carbon loss that was greatest 10–20 years after harvesting, and to slowly recover thereafter (Jiang et al. 2002). A 30-year period of post-harvest soil carbon loss was observed in Nova Scotia red spruce (*Picea rubens*) forests, including from the deep mineral soil (Diochon et al. 2009).

A growing body of research, however, suggests that post-harvest respiration is not as important in the carbon budget as Covington (1981) suggested. A critical re-visit of his study suggested that the loss of organic mass from the forest floor after harvest was due to intermixing into the mineral soil, not increased decomposition (Yanai et al. 2003). If this is true, then the carbon consequences of harvesting are quite different, since organic carbon incorporated into the mineral soil may actually increase total carbon sequestration on the site.

Several comprehensive reviews of harvest effects on soil carbon also indicate limited impact. Depending on the level of slash input and organic matter incorporation into the mineral soil, harvests can result in slightly negative or slightly positive, or often no changes in soil carbon (Johnson 1992; Johnson and Curtis 2001). Conversion of old-growth *Picea* forests in British Columbia to young plantations reduced litter carbon stocks but left mineral soil carbon unaffected (Fredeen et al. 2007). Little or no net loss of forest floor weight was associated with clearcutting or partial cutting in Canadian boreal mixedwoods, perhaps due to rapid return to pre-treatment light and moisture conditions after prolific trembling aspen (*Populus tremuloides*) sprouting (Lee et al. 2002). In both Ontario northern hardwoods (Laporte et al. 2003) and Ozark oak forests (Edwards and Ross-Todd 1983; Ponder 2005; Li et al. 2007), uneven-aged management led to increased soil carbon levels, and clearcutting resulted in no significant change, compared to controls. Rates of both root respiration and microbial respiration may decline after harvest due to tree removal and soil compaction (Laporte et al. 2003). Where increased efflux has been observed, it tends to be small and limited to the uppermost soil layer (such as in a Chilean *Nothofagus pumilio* shelterwood (Klein et al. 2008)), and recovers to pre-harvest conditions after only a few years (aspen clearcuts in Ontario, Canada (Weber 1990)).

Johnson and Curtis (2001) hypothesized that whole tree harvesting could potentially result in soil carbon losses because of the high rates of biomass removal from the site. However, field studies in northern New Hampshire and Maine indicate that this practice results in no reduction in forest floor mass or soil carbon pool relative to uncut areas (Huntington and Ryan 1990; McLaughlin and Philips 2006). Some research suggests that the long-term consequences of management on soil carbon pools will be stronger than the short-term. A 300 year model of Canadian boreal forests shows a consistent decline in soil carbon in managed forests (Seely et al. 2002). Multi-rotation monitoring of managed forests will be necessary to assess the rigor of such models.

As the above studies indicate, there is significant evidence to show that if there is any soil carbon loss following a harvest, it is a short-term component of a site's carbon budget. Mineral soil carbon is usually not affected by harvest, and the loss from litter layers can be offset by slash additions. If the impact on soil carbon is indeed minor, then intensive pre- and post-harvest measurement of soil carbon pools may not be necessary. One of the main criticisms of making soil carbon measurements a low priority is that the research supporting it rarely involves measurement of deep soil carbon. One of the few studies to do so (in a red spruce chronosequence in Nova Scotia) found that younger post-harvest stands had significantly lower carbon storage at the 35–50 cm soil depth (Diochon et al. 2009). Before the conclusion can be made that soil carbon pools are not significantly affected by harvesting, greater attention must be paid to these deep soil layers. A single meta-analysis of impacts of harvesting on mineral soil carbon reveals that soil taxonomy perhaps provides the greatest explanation for susceptibility to mineral soil carbon loss, with ultisols and inceptisols showing a net loss of 7% and 13% respectively, while spodosols and alfisols remained unchanged (Nave et al. 2010). In the same review surface litter horizons were much more sensitive with losses amounting to 30% (hardwood litter had greater carbon loss than coniferous); clearly more studies are needed to substantiate these claims (Nave et al. 2010).

If all the carbon pools, inputs and outputs are considered together, it appears that clearcut stands are carbon sources for the first decade after harvest (thanks to transient increases in respiration), after which they switch to sinks. This pattern holds for boreal forests in British Columbia (Fredeen et al. 2007), Saskatchewan (Howard et al. 2004) and Finland (Kolari et al. 2004), but its applicability in temperate zones is not as clear. Partial regeneration harvests (shelterwoods, selection) appear more site and soil specific. For example many second growth even-aged forests in New England can be managed to increase structural complexity and hence stored carbon by retaining older and larger trees (reserves) within the stand during a regeneration harvest (Keeton 2006; Evans and Perschel 2009; Ashton et al. in press)

4.4.3 Treatment of Harvest Residues

The addition of harvest residues to the litter and soil layers is an important factor in mitigating initial carbon loss from harvested forests. This might suggest a negative carbon influence from removing these residues (and natural litterfall) for utilization, fuel reduction, or site preparation. However, research is mixed. Balboa-Murias et al. (2006) found that logging residues contained 11% of the total biomass carbon stored across a rotation in Spanish radiata pine (*Pinus radiata*) and *P. pinaster* plantations. They thus concluded that residue harvest for biomass burning (a common practice in Spanish forests) would result in reduced ecosystem carbon storage. Piling and burning slash in California clearcuts resulted in soil carbon loss (Black and Harden 1995). Removing harvest residues alone from New Zealand *P. radiata* plantations did not significantly alter soil carbon levels, but removing residuals and the forest floor (i.e. accumulated litterfall) did. In addition, a pattern of increasing soil carbon stocks with increasing residue retention was observed (Jones et al. 2008). In oak forests of Missouri, there was no significant increase in soil respiration between whole-tree harvest and whole-tree harvest + forest floor removal, and both had lower respiration than the control (Ponder 2005). In Australian *Eucalyptus* forests, residue retention had minimal impact on soil carbon levels, but may have some influence if practiced across multiple rotations (Mendham et al. 2003).

It appears that removing logging slash from harvested sites reduces the litter carbon pool, which is important in some forest types. But unless the natural litterfall is also reduced, residue removal has limited impact on soil carbon levels. Moreover the overall carbon impact of biomass removal depends in large part on its utilization such as replacement for fossil fuels (Evans and Finkral 2009) and greenhouse gases produced by its decomposition (Chen et al. 2010).

4.4.4 Changing Rotation Length

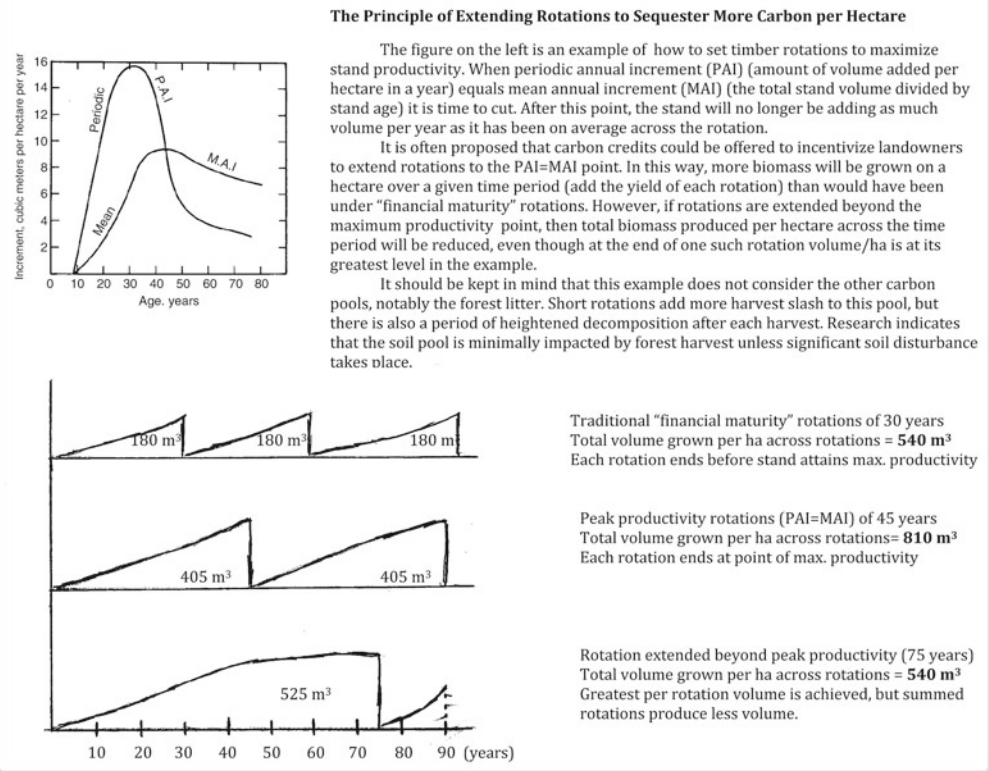
Many forests in the temperate and boreal zones are managed on rotations far shorter than the potential age of the species present. Often these rotations are so short that the maximum biomass productivity possible on the site (the “ceiling”) is never reached. In a broad review of forest management effects on carbon storage, Cooper (1983) found that, on average, stands managed for maximum sustained yield store only 1/3 of the carbon stored in unmanaged, late successional forests.

Management for a financially optimal rotation results in an even smaller storage.

Research has shown the possibility of creating carbon additionality (in comparison to business-as-usual managed forests) by increasing rotation length. In Chinese boreal forests, Jiang et al. (2002) modeled a variety of rotation lengths and found that 30-year rotations stored only 12% as much carbon as 200-year rotations. In Europe, rotation modeling of spruce and pine forests showed increased carbon storage with increased rotation. This is especially true where stands retain high net primary productivity (NPP) rates even at extended rotations, such as pine plantations in northern Spain (Kaipainen et al. 2004). Further research in Spain supported this finding, although the authors noted that mean annual carbon uptake eventually will decline with increasing rotation as trees become less productive (Balboa-Murias et al. 2006). Jandl et al. (2007) found that lengthening rotations would increase carbon storage until stands reached an advanced developmental stage in which biomass actually began to decline (as observed in some old-growth forests).

As is often the case, the impact of rotation length on soil carbon is complicated. One Finnish study found that soil organic matter was maximized with shorter rotations, because of increased slash inputs to the litter and soil layers (Pussinen et al. 2002). Lengthening rotations in models of wood production in Finland resulted in greater carbon storage when the increase in biomass carbon exceeded the decrease in soil organic matter. This occurred in the case of Scots pine, but not for Norway spruce, suggesting that short rotations are more carbon-positive for the latter

Insert 4. The Principle of Extending Rotations to Sequester More Carbon Per Hectare



The Principle of Extending Rotations to Sequester More Carbon per Hectare

The figure on the left is an example of how to set timber rotations to maximize stand productivity. When periodic annual increment (PAI) (amount of volume added per hectare in a year) equals mean annual increment (MAI) (the total stand volume divided by stand age) it is time to cut. After this point, the stand will no longer be adding as much volume per year as it has been on average across the rotation.

It is often proposed that carbon credits could be offered to incentivize landowners to extend rotations to the PAI=MAI point. In this way, more biomass will be grown on a hectare over a given time period (add the yield of each rotation) than would have been under "financial maturity" rotations. However, if rotations are extended beyond the maximum productivity point, then total biomass produced per hectare across the time period will be reduced, even though at the end of one such rotation volume/ha is at its greatest level in the example.

It should be kept in mind that this example does not consider the other carbon pools, notably the forest litter. Short rotations add more harvest slash to this pool, but there is also a period of heightened decomposition after each harvest. Research indicates that the soil pool is minimally impacted by forest harvest unless significant soil disturbance takes place.

Traditional "financial maturity" rotations of 30 years
Total volume grown per ha across rotations = **540 m³**
Each rotation ends before stand attains max. productivity

Peak productivity rotations (PAI=MAI) of 45 years
Total volume grown per ha across rotations= **810 m³**
Each rotation ends at point of max. productivity

Rotation extended beyond peak productivity (75 years)
Total volume grown per ha across rotations = **540 m³**
Greatest per rotation volume is achieved, but summed rotations produce less volume.

species (Liski et al. 2001). This must be tempered, however, by the increased fossil fuel emissions associated with short-rotation forestry (Liski et al. 2001).

The principle behind lengthening rotations is to bring stands closer to the advanced ages at which maximum biomass is attained. By this same principle, forests that are already in these stages (for instance, old-growth) should be maintained. Harmon et al. (1990) considered the carbon consequences of the conversion of old-growth forests in the Pacific Northwest to managed production forests, finding that it caused a reduction in carbon storage that extended for 250 years, and could probably never be made up for. If forests in this region were managed with rotations of 50, 75 and 100 years, the carbon stored would be at most 38%, 44% and 51%, respectively, of that stored in old-growth (Harmon et al. 1990). Tang et al. (2009) predicted a similar long-term loss in ecosystem carbon with the conversion of Michigan northern hardwoods to younger stand structures. Managing red spruce on 60 year rotations in Nova Scotia would result in the loss of 42% of soil carbon relative to old-growth and 26% relative to 80 year rotations (Diochon et al. 2009). Managed *Eucalyptus* forests in Australia contain only 60% of the aboveground vegetative carbon stored in old-growth.

The key explanation of this discrepancy is the dearth of large (>100 cm in diameter) trees in managed stands. In old-growth rainforest/eucalyptus stands in New South Wales, Australia, such trees make up only 18% of the stems >20 cm, but contain 54% of the vegetative carbon (Roxburgh et al. 2006). These studies suggest, at the least, that when old-growth forests already exist, their maintenance is optimal for carbon sequestration.

5 Management and Policy Implications

5.1 Recommendations for Land Managers

- Relatively few forest management practices can demonstrate true carbon additionality. Afforestation/reforestation usually increases a site's carbon sequestration, unless it results in a significant release of soil carbon (i.e. through intensive site preparation or the oxidation of peat soils). The impact of afforestation/reforestation on soil carbon pools must be carefully monitored.
- Thinning causes a reduction of the vegetative carbon stored on-site, which recovers over a matter of decades (depending on thinning intensity and tree vigor). Thinning's impact on soil carbon appears very limited, as inputs of slash and reduced root respiration seem to make up for reduced litterfall and increased microbial respiration.
- Resiliency treatments (such as fuels reduction thinning and prescribed fire) result in lowered carbon storage on-site and some carbon release from decomposition and combustion. However, they help produce forests that are significantly less susceptible to stand-replacing disturbance (with accompanying carbon releases). Essentially, forest managers using these treatments accept less than maximum short-term carbon storage to ensure long-term and more secure storage.
- Fertilization treatments that improve the nutrient conditions limiting plant growth can increase the vegetative carbon pool (particularly on marginal soils), and increase the soil carbon pool by reducing root and microbial respiration. This must be tempered by consideration of the carbon footprint of fertilizer production, which can match or exceed the additional carbon sequestration.
- Draining of saturated peat soils and subsequent afforestation can cause either a net carbon loss or gain, depending on whether increased tree growth and litterfall and decreased methane release outweigh the increase in respiration from oxidized peat. This may in turn be dependent on the extent to which drainage lowers the peatland water table. Research from drained lands in Finland and the British Isles indicates that net carbon sequestration is possible when the water table remains relatively high after drainage.
- Regeneration harvests significantly reduce the carbon stored on-site, especially even-aged

treatments such as clearcutting. The amount of stored carbon may not rebound for many decades (or centuries, if the pre-harvest stand was in old-growth condition), but the annual rate of carbon uptake will be greater in the regenerating stand. Harvested stands often are net sources of carbon for the first 10–30 years, because of increased litter and soil respiration. They then become net sinks as vegetative growth and litter accumulation exceed respiration.
- Removing harvest residues (slash) for biomass utilization, to reduce fuel levels or to prepare the site for planting, directly reduces the litter carbon pool. The impact on soil carbon is less clear. Treatments that only reduce slash do not result in significant soil carbon loss (over one rotation), but loss occurs if the forest floor (natural litter accumulation) is removed as well.
- Managing stands for maximum sustained yield or financially optimum rotation can result in non-optimal carbon storage. Such rotations are often too short to allow the stand to attain maximum biomass. As such, it is often possible to increase carbon sequestration by extending rotations. This is particularly true on productive sites where high rates of NPP can be sustained through longer rotations. There is a point of diminishing returns, though, when rotations are extended beyond the age of maximum biomass productivity. At some point, it may be possible to store more carbon in a series of short rotations (that maintains the stand in a young, productive stage) than a single longer rotation.
- If old forests *already exist,* however, maintaining them as old forests maximizes carbon storage. Old forests, especially on productive sites, often have very large pools of vegetative carbon in comparison to forests managed on shorter rotations. Soil and litter pools may also be quite large in old-growth forests, and in the boreal, the bryophyte pool as well. The conversion of old-growth to managed forests likely results in a loss of ecosystem carbon that cannot easily be regained.

5.2 Recommendations for Policy Makers

- The concept of carbon additionality is central to carbon credit and offset schemes. It is difficult to demonstrate additionality in most forest management practices. By its nature, forest management often causes reductions in carbon stocks, especially from the vegetative pool. But a contribution can still be made to climate change mitigation by adjusting these practices so as to *minimize carbon release* as opposed to *maximizing carbon sequestration.* The former idea is gaining traction through such mechanisms as offsets for reduced deforestation/degradation and reduced impact logging. If boreal and temperate forests are to be included in a carbon credit and offsets scheme, it will likely be necessary to recognize such contributions, which are potentially more feasible than “traditional” carbon additionality.
- If policy makers choose to include such “reduced carbon release” practices in a credit/offset scheme, they will need to set a baseline that allows these practices to demonstrate additionality. If the baseline is a natural, unmanaged forest, then most forest practices will always appear carbon-negative. But if the baseline is a “business-as-usual” managed forest, then such practices will constitute a creditable improvement over the baseline. Setting baselines is not a purely scientific process; it is an act of policy that determines which forest management activities will be incentivized.
- The practice of extending rotations offers a straightforward biological means of increasing carbon sequestration in existing forests, and thus has become a focus for forest managers participating in carbon offset markets. It has been suggested that carbon offset credits can be used to produce a large-scale dividend of additional carbon sequestration by subsidizing landowners to extend rotations until peak stand productivity (in silvicultural terms, when periodic annual increment and mean annual increment are equal) (Wayburn 2009). In this way, carbon “density” per unit area will

be increased by allowing forests to more closely approach their natural productive potential.

- The well-known market externality of "leakage" complicates the implementation of concept such as extending rotations. If revenues from carbon credits motivate enough landowners to extend rotations, then demand for wood shifts elsewhere. The landowners may well plan to harvest the same (or greater) volume several decades from now, but that does nothing to change the current demand for wood. Mills will be forced to increase the price they pay for roundwood, which will likely motivate landowners not participating in carbon sequestration activities to cut and sell more wood than they otherwise would have (and perhaps *earlier* in the rotation than they planned). Thus, while some landowners delay harvesting in order to accumulate more carbon per forested acre, other landowners will accelerate harvest to fill the gap, neutralizing net carbon gains.
- Another important policy factor is whether to consider forest products as a carbon pool. The choice could well determine whether or not practices like thinning are positive, neutral or negative from a carbon sequestration perspective. If the carbon contained in forest products is "sequestered," then a great many more forestry projects would be eligible for carbon credits and offsets than if that carbon is "released." The designers of offset systems will need to balance the increased measurement and documentation burden of including a forest products carbon pool with the potential to include more projects.
- Many forest management practices have a minimal impact on the soil carbon pool, which is the most difficult to measure. Thus, it may be possible that offsets involving certain forestry practices could go forward without strict quantification of this pool. This would considerably reduce measurement cost. As a rule, quantification would likely be least vital when the practice in question results in minimal soil disturbance.

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