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Review

# Mitigation, adaptation, and climate change: results from recent research on US timber markets

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**Abstract**

This paper reviews recent studies that have addressed how US timber markets may adapt to climate change, and how US forests could be used to mitigate potential climate change. The studies are discussed in light of the ecological and economic assumptions used to estimate adaptation. Estimates of both economic impacts and carbon sequestration costs depend heavily on the assumptions and methods used, although some general conclusions can be drawn. Studies of economic impacts suggest that average market effects in the United States may range from + $1.3 to + 7.4 billion per year by the middle of the next century. Estimates of the cost of sequestering carbon have generally increased over the last 10 years, with a current range of < 1–73 million metric t per year of additional sequestration from afforestation projects costing $5–66 per metric t. Estimates of the potential for alternative methods for carbon sequestration, such as product markets and recycling, are as large as afforestation estimates, with up to 50 million metric tons per year of additional storage considered possible. Cost estimates have not been developed for these alternative methods, however. © Published by Elsevier Science Ireland Ltd.

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## 1. Introduction

Over the past 15 years, links among forests, human activity, and potential climate change have been heavily researched. This paper assesses a small part of this large research area by reviewing a number of economic studies that have investigated adaptation and mitigation in forestry. Studies on adaptation have focused mostly on exploring the relationship between the ecological impacts of climate change and timber markets. Given that recent assessments suggest that 'adaptation options for ecosystems are limited, and their effectiveness is uncertain (Watson et al., 1998),' a closer examination of existing economic studies may provide additional insights into adaptation processes. We also assess economic studies that have investigated the possibility of using forests to store additional carbon. Mitigation represents one of the most important links among humans, forests, and climate change through potential effects on the global carbon cycle.

Although numerous studies have investigated adaptation and mitigation throughout the world, this review focuses mainly on research conducted in the United States. One reason is that there has been significant effort in the United States to link ecological models to economic models in order to assess adaptation. In addition, most ecosystems in the US are dominated by humans and understanding how humans adapt in these ecosystems is crucially important for understanding potential impacts not only on industrial wood markets, but also on biodiversity, carbon storage, and other services from forests. These other services are not specifically considered in this paper, and we note that additional research could be undertaken to benefit the policy process.

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Unlike previous reviews of the economic impacts of climate change on timber markets (see, for example, Winnett, 1998), this paper focuses more explicitly on the models used to generate the results, and how the underlying economic and ecological modeling assumptions potentially affect the results of the studies. Although differences in assumptions allow researchers to address specific questions, they necessarily make comparisons more difficult. This paper attempts to make more explicit the basic assumptions used by the modelers in order to aid analysis and discussion of similarities, differences, and implications. In addition to examining results, reviewing differences among studies can help to identify useful issues for future forestry research.

Our review of the mitigation literature also differs from previous assessments (see Sedjo et al., 1995). The economic modeling technology has changed significantly since this earlier review, and we believe that these improvements have implications for estimates of the costs of sequestration. First, a wide range of economic models have been used for estimating the costs of afforestation. These studies are focused on the US because the models are data intensive, and the data are not always available to apply the techniques elsewhere. However, these studies provide insights that may be instructive for individuals attempting to estimate costs of afforestation elsewhere. Second, additional studies have investigated alternative policy options for carbon sequestration, including forest fire management, storage in product markets, recycling, and timber management. Although cost estimates have not typically been made in these studies, they provide information on the potential for the activity to increase storage. Finally, we review a number of policy issues associated with mitigation. For example, the total costs of afforestation programs are a function not only of the direct land and planting costs — the issue that most studies have addressed — but they are also a function of the methods used to implement programs. Most studies to date have ignored these issues, and this review addresses potential implications.

## 2. Economic impacts of climate change

To date, most economic studies of climate change impacts on timber markets use a series of models that are linked linearly, flowing from general circulation models (GCMs) to ecological models to economic models. Nearly all published studies begin with equilibrium, doubled $CO_2$, climate change predictions from GCMs. Ecological models use these results to predict the equilibrium ecological consequences. Because most economic models simulate market activity on an annual or decadal time step for a number of projection years, the ecological and economic modelers must develop transient scenarios of the ecological change. Transient scenarios are then introduced into the economic models and economic impacts in different time periods are estimated. Importantly, the transient scenarios used by economists to date have been determined exogenously from the equilibrium ecological models. Over the past few years, both the GCM and ecological models have begun to produce transient scenarios of climate change and the attendant transient ecological impacts (see, for example, Daly et al., 2000), although there are no published economic estimates to date that use transient ecological models.

### 2.1. Ecological modeling assumptions used in economic analyses

A wide range of ecological modeling assumptions have been used to estimate impacts of climate change on timber markets. For the most part, economists have used large-scale, spatial ecological results, such as those contained in VEMAP Participants (1995), rather than stand-specific models. Stand-level simulation models, such as GAP models, typically predict how forest stands respond to gaps in the forest canopy (see Shugart et al., 1986, for a discussion of the early application of GAP models to climate change questions). The main reason economists have focused on using spatial models rather than stand models is that estimating economic impacts and adaptation in markets requires economists to model the response of timber prices to climate change. Because prices are set in markets where wood products are traded across regions (either within the US or internationally), understanding climate change impacts across large areas or numerous regions is important for ultimately measuring the price response. A related reason is that spatial ecological results can provide insights into how different regions gain or lose comparative advantage in timber production and supply during climate change.

Economic studies have focused on two effects of climate change on forests. First, a number of studies have used the results of *biogeochemical* cycle models (Parton et al., 1988; Running and Coughland, 1988; Running and Gower, 1991; Melillo et al., 1993) to explore how forested ecosystems will respond to climate change and altered $CO_2$ levels. Biogeochemical cycle models predict a number of ecological variables, but the variable most frequently used as input into economic models is net primary productivity (NPP). NPP is used to predict changes in annual growth of trees. This change is linked to economic models by adjusting existing annual growth functions for merchantable timber proportionally to predicted changes in NPP. These changes can be quite substantial. For example, under a range of three GCM models and three biogeochemical cycle models, VEMAP Participants (1995) predicts that average NPP across forested ecosystems in the US may change by −6.5 to +17.0% without considering carbon fertilization effects and +9.4 to +34.6% with carbon fertilization effects included.

To a lesser extent, economic models have attempted to capture changes in the distribution of species. The link between climate and species distribution has long been recognized (Koppen, 1884; Holdridge, 1947; Thornthwaite, 1948), and *biogeographical* models have been used to show how climate change and altered $CO_2$ may alter the future distribution of vegetation (Box, 1981; Emanuel et al., 1985; Prentice et al., 1992; Neilson et al., 1992; Neilson and Marks, 1994; VEMAP Participants, 1995). As with the predicted changes in NPP, these changes may be substantial for the United States. For instance, using three GCM models and three biogeographical models, VEMAP Participants (1995) predicts that total forest area in the United States may change from −84 to +7% without carbon fertilization effects, and −13 to +23% with carbon fertilization effects.

Both the biogeographical and biogeochemical models provide significant regional detail that can be used by economic models. Although this allows economists to measure regional impacts, there are costs. One of the costs is that the GCM model results are considered to be less certain for predicting regional impacts than for predicting global impacts (Watson et al., 1998). Another is that ecological and economic modelers both must make assumptions from model results earlier in the chain of models described above. For instance, ecological modelers typically interpolate results from GCM models at grid cells greater than two degrees to the half-degree size for ecological predictions. Economic modelers then re-aggregate these results for use in economic models. The level of aggregation varies across the models. For example, Sohngen and Mendelsohn (1998) aggregate to four timber types within the US, whereas Joyce et al. (1995) aggregate to 55 timber types, as defined by the USDA Forest Service. Rather than measuring economic impacts in particular grid cells, economic models use average changes across aggregations of grid cells as input.

There are a number of reasons for spatial aggregation in economic analysis. First, supply functions in timber market models are based on forest inventory data that contain information on the age class distribution of trees. Age class information in the United States is more statistically reliable at the regional level than at the half-degree grid cell level. Note that half-degree grid cells are approximately the size of US counties east of the Mississippi river; so having age class information for these grid cells would require abundant forest age information at the county level. These data are generally not available in a statistically reliable form for all counties. Second, economic modelers have addressed regional level effects in order to capture national or global price effects. These price effects are crucially important for measuring adaptation and economic impacts to consumers and producers. Because harvests in any particular region are linked by prices to ecological conditions, harvests, and demand in other regions, developing models that link half-degree grid cells rather than regions would complicate dramatically the solving of models. Economic models could be developed to focus on the effects of ecological changes in half-degree grid cells, but these models would likely miss the price effects caused by climate change elsewhere.

Another limitation is that the ecological models used in economic analyses to date predict only equilibrium conditions. These equilibrium conditions may not occur for 100–200 years into the future, if ever. Policy analysis is largely interested in the near term (10–50 years) consequences of climate change. Current efforts with many of the models used in the VEMAP process are focusing on determining the transient response of forests to a slowly changing climate (see for example, Daly et al., 2000), but it will take some time for economic analyses to fully incorporate these transient responses.

A final limitation is that existing economic studies that have considered species movements from one place to another (for example, Sohngen and Mendelsohn, 1998) rely on early one-way model links from biogeographical to biogeochemical cycle models (see VEMAP Participants, 1995). Changes in species distribution could affect NPP, but in the VEMAP study, NPP did not have an effect on the biogeographical distribution of species. This could be an important limitation of earlier estimates. More recent efforts (such as the work of Haxeltine and Prentice, 1996; Daly et al., 2000) have explicitly linked biogeographical and biogeochemical effects.

### 2.2. Economic modeling of climate change impacts

Economic analysis that captures the human response to ecological change is the fourth or fifth step in the chain of models and assumptions used to estimate economic impacts and adaptation. Thus, the economic results contain not only economic uncertainty, but also uncertainty from earlier results in the chain of modeling. Most economic studies address the uncertainty from earlier models by using several combinations of both climate and ecological models, which capture a range of results. A related issue is that both GCM's and ecological models are evolving quite rapidly. One can expect that the economic results will evolve as well as modelers include newer results, such as those from recent transient GCM and ecological models.

Existing economic studies, nevertheless, provide two key insights into adaptation. First, they provide information on the potential scale of the economic effects in timber markets based on a set of potential climate and ecological effects. This information may be useful for integrated assessment efforts focused on aggregating potential effects across regions and across sectors, and in sensitivity analyses of key determinants or factors in the chain of models. Second, the economic studies provide a number of useful insights into the process of adaptation in land and product management.

One of the difficulties involved with comparing individual studies of timber market impacts is that several types of economic models have been used. The general differences between the four most commonly used economic models reviewed here are presented in Table 1, and are discussed in detail in Sohngen (1997), Sohngen and Sedjo (1998). From the perspective of modeling climate change, two differences appear to be most important. First, the models treat capital investments in timber differently. The forest and agricultural sector optimization model (FASOM) and the model used in Sohngen and Mendelsohn (1998) are forward looking, which means that investments in timberland management are determined by future price expectations. Future price expectations are assumed to be correct on average. TAMM/NAPAP/ATLAS/AREACHANGE and CGTM, on the other hand, assume exogenous time paths for timberland investments.

Second, the models have different methods for projecting timber supply as inventory ages. FASOM and Sohngen and Mendelsohn (1998), for example, allocate the harvest age endogenously. In FASOM, the oldest age classes are generally harvested first, and in Sohngen and Mendelsohn (1998), the oldest timber is always harvested first. The specific age in both models, however, is determined endogenously, depending on climate impacts and prices. In both models, the inventory of oldest age classes has the most important effect on timber supply. The supply functions in TAMM/NAPAP/ATLAS/AREACHANGE and CINTRAFOR global trade model (CGTM) rely on the entire merchantable inventory of timber. As shown in Sohngen and Sedjo (1998), this difference can have large effects on the resulting price and inventory path depending on the underlying ecological changes. Although these differences are important, it is difficult to assess their implications for the economic results relative to differences in ecological assumptions that have been used by the modelers to date. For instance, Sohngen and Sedjo (1998) show that the largest differences among model responses occur when there is a loss of stock, such as through dieback, but only one study reviewed here (Sohngen and Mendelsohn, 1998) explores dieback phenomena related to climate change. Differences between the economic model outcomes would likely become more apparent if age-specific effects were modeled. To date ecological modelers have not provided these age-specific impacts, so most economic studies have simply assumed that the ecological effects are the same for all age classes of trees — an assumption that dampens the effect of differences in economic methods on the results.

### 2.3. Assessment of adaptation and impacts in individual studies

This section takes a closer look at several specific studies, and the economic impacts and adaptation predicted by each. Because the underlying models in the studies have made a variety of assumptions about

Table 1
General economic characteristics of four timber market models that have been used for climate analysis

| | TAMM/NAPAP/ATLAS/AREACHANGE used by Joyce et al. (1995) | CGTM used by Perez-Garcia et al. (1997) | FASOM Used by Burton et al. (1998) | Sohngen and Mendelsohn (1998) |
|---|---|---|---|---|
| *Timber model attributes* | | | | |
| Economic theory | Spatial equilibrium | Spatial equilibrium | Dynamic optimization | Dynamic optimization |
| Projection method | Static simulation | Static simulation | Optimal control | Optimal control |
| Harvest age | Exogenously determined by region | Exogenously determined by region | Endogenously determined — generally oldest first | Endogenously determined — oldest first |
| Global scope | US and Canada | Global | United States | US softwoods |
| Regions | ~8 | ~40 | ~9 | 4 |
| Tracks regional trade within the US | Yes | Yes | Yes | No |
| Market structure | End product and log | End product and log | Log | Softwood stumpage |
| Capital adjustment | Depends on current profit margins | Depends on current profit margins | Depends on future price expectations | Depends on future price expectations |
| Timber inventory | Age-delimited | Age-delimited | Age-delimited | Age-delimited |
| Land/timber management | Exogenous projection | Exogenous projection | Determined by model | Determined by model |

Table 2
General characteristics of climate change and ecological models used by four studies addressing the impacts of climate change on US timber markets

| | Joyce et al. (1995) | Perez-Garcia et al. (1997) | Burton et al. (1998) | Sohngen and Mendelsohn (1998) |
|---|---|---|---|---|
| *Climate and ecological model attributes* | | | | |
| Climate scenarios | 4 | 4 | None Explicit | 2 |
| Biogeochemical models | 1 | 1 | None Explicit | 3 |
| Forest productivity | Affects growth | Affects growth | Affects growth | Affects growth |
| Biogeographical models | None | None | None | 3 |
| Forest species migration | None | None | None | Dieback and regeneration |
| Dynamic climate change | Linear | Linear | Compounded | Linear |
| Years to doubling | 75 | 75 | 60 | 70 |
| Years in model run | 50 | 50 | 60 | 150 |

climate and ecological effects, including using different climate models, different ecological models, and different methods for developing transient pathways from equilibrium results, the general differences are highlighted in Table 2. Although these differences make it more difficult to compare the specific results, it is still possible to make some general conclusions about potential adaptation and economic impacts.

Binkley (1988) appears to have been the first author to explicitly link the ecological effects of climate change with a timber model, although he considered only changes in boreal forests, and he did not predict market effects in the United States. More recently, Haynes et al. (1995), Joyce et al. (1995) linked the TEM biogeochemical cycle model (Melillo et al., 1993) to the TAMM/NAPAP/ATLAS/AREACHANGE model to project how timber markets in the United States adapt to changes in ecosystem production. The ecological changes from TEM imply increased NPP in general, increased timber inventories, and larger timber harvests in the future (Table 3). The researchers do not consider changes in the distribution of timber types as predicted by biogeographical models.

The TAMM/NAPAP/ATLAS/AREACHANGE model adjusts manufacturing capacity annually in line with relative profit margins across regions. Although the ecological models predict that the North gains comparative advantage over the South in growing trees, economic adaptation is limited in this modeling approach by the movement north of timber processing capital. Current timber production in the North is an order of magnitude lower than that in the South, and adding capacity to pulp and paper production in the North would require large investments that could cost billions of dollars. Haynes et al. (1995), Joyce et al. (1995) thus predict that production remains high in the South throughout the 50-year projection period because that region has a large proportion of the nation's existing capital for producing timber products, and that capital is not projected to shift rapidly northward during climate change. A limitation of the version of the TAMM/NAPAP/ATLAS/AREACHANGE modeling systems employed by Haynes et al. (1995), Joyce et al. (1995) is that timber mortality and salvage are not explicitly modeled. This is an area of both ecological and economic uncertainty, but if mortality is predicted to differ from the past, the TAMM/NAPAP/ATLAS/AREACHANGE modeling system is not designed to seamlessly model such harvests.

Perez-Garcia et al. (1997) also link the TEM biogeochemical cycle model to an economic model, but they use the same CGTM that was used by Binkley (1988). Unlike the earlier study by Binkley, Perez-Garcia et al. model global effects, not just effects in one region. TEM predicts increased global NPP, although some regions gain and some regions lose. These lead to changes in relative timber growth rates, and consequently to changes in regional timber inventories. The global effect is to increase timber inventories and reduce prices. This benefits consumers and harms landowners. It also affects bi-lateral trade flows, although those are not discussed in the paper. As with the TAMM/NAPAP/ATLAS/AREACHANGE model, the adaptive capacity of the wood-processing sector in CGTM is limited. For the United States, the authors project that lumber and plywood production increases, although pulpwood production decreases under some scenarios. US consumers and mill owners gain from lower prices, while landowners are less well off. The annual net effects in 2040 are projected to be approximately + $1.3 billion per year (1990 US dollars) for the United States.

Sohngen and Mendelsohn (1998) explore 36 combinations of climate, biogeochemical, and biogeographical changes (Table 2). They focus on adaptation in forestland management under the assumption that wood-processing capital shifts to regions with comparative advantage in timber growth. Forest area change is captured with either a dieback scenario (King and Neilson, 1992), or a less dramatic regeneration scenario. Salvage in the dieback scenarios ranges from 50 to 75%, depending on management intensity. Their model considers only softwoods, and these markets are projected to adapt to the short-term effects of climate change by reducing prices, salvaging dead and dying timber, and replanting new species that are favorably adapted to the new climate. Softwood market value is expected to increase by 2–6% in all scenarios considered. The average net consumer and producer effects are estimated to be a + $20 billion over the entire 150-year projection period. Estimates of annual effects, similar to those presented by Perez-Garcia et al. (1997), suggest average benefits in 2060 on the order of + $7.4 billion per year (1990 US dollars), with a range of + $0.1 to + 11.9 billion per year.

Burton et al. (1998) used FASOM (Adams et al., 1996; Alig et al., 1998) to determine how 50% increases or decreases in growth rates for northern and southern US species would affect US timber markets. Their yield changes are not based on underlying climate or ecological models, but by explicit assumption in lieu of uncertain estimates from existing GCM and ecological production models. Two interesting implications for adaptation emerge from this study. First, if northern and southern species have similar increases in timber growth rates, comparative advantage shifts from the south to the north. The rate of shift is apparently faster than in the TAMM/NAPAP/ATLAS/AREACHANGE model due to the assumption that markets are forward looking and there are no limitations on the adaptive capacity of timber companies. Second, similar increases in growth rates favor sawtimber over pulp. The given reason for this is that sawtimber becomes relatively less expensive to grow, and therefore, gains some new advantages when growth rates are increased. In terms of economic impacts, the specific results are not provided, however, they can be inferred from graphs in the paper. Under the 50% national decline in growth rates, net consumer and producer effects for 2050 are − $230 billion per year (1990 US dollars), and under the 50% national increase in growth rates, these effects are + $60 billion per year. When growth declines 50% in the south, but increase 50% throughout the rest of the country, the net effects are − $5 billion per year, indicating the relative importance of the South in US timber markets.

### 2.4. Implications and limitations

Each of the models considered above suggests that markets will adapt to changing prices, although fundamentally different processes lead to the adaptation shown by the models. In Joyce et al. (1995), Mills and Haynes (1995), Perez-Garcia et al. (1997), changes in the relative growth rates of different species lead to adjustments in inventories for hardwoods and softwoods, and changes in relative prices. Markets then adapt by adjusting demand for hardwoods and softwoods, and by adjusting capital investments in processing machinery for hardwoods and softwoods. Markets thus balance inventories, prices, and the use of alternative inputs in the end product market. By responding to

Table 3
Comparison of US results from four studies of the economic effects of climate change on timber markets

| | Joyce et al. (1995) | Perez-Garcia et al. (1997) (US only) | Burton et al. (1998) | Sohngen and Mendelsohn (1998) |
|---|---|---|---|---|
| NPP/TIMBER yield (long-term equilibrium) | Average ≈ +15% (range, −7 to +43%)[a] | Average ≈ +10% (range, 0 to +30%[a,b] | Across the board effects (−50 or +50%) | Average ≈ +7% (range, −35 to +50%)[a] |
| Loss of initial forest area | None modeled | None modeled | None modeled | Softwood area loss by 2060 (25–46%) |
| Forest area | No change relative to baseline | No change relative to baseline | Measured endogenously | Softwood area gain by 2140 (+6 to +38%) |
| Inventories | Increase | Increase | Increases or decreases | Long-term Increase |
| Timber prices compared with base | Lower due to increase in US inventory | Lower due to increase in global inventory | Increase (decrease) when yield decreases (increases) | Lower due to salvage and long-term increase in inventory |
| *Market welfare effects* | | | | |
| General response | Not given | Consumers gain producers lose net effect positive | Increases or decreases | Net effect positive |
| Total net welfare effect (billion 1990 US$) | Not given | Not given | Not given | Average, +$20; range, +$0.9 to +33.2 |
| Annual net welfare effect (billion 1990 US$ per year) | Not given | Avearge in 2040 = +$1.3 | Range in 2050 = −$230 to +60 | Average in 2060 = +7.4 Range = +$0.1 to +11.9 |

[a] Average effects are based on data presented in the papers.
[b] Estimates are only for US results presented in Perez-Garcia et al. (1997).

prices, markets have a large capacity to make 'small' adjustments to minimize the overall effects. Sohngen and Mendelsohn (1998) focus on how changes in species distribution, potential dieback and salvage, and changes in growth rates alter harvesting and replanting behavior for different species. Although dieback may have significant ecological consequences, these results suggest that dieback provides powerful incentives for markets to adapt by harvesting stocks earlier than otherwise, or by replanting new species that are adapted to a new climate. Sensitivity analysis by Sohngen and Mendelsohn (1998) suggest that welfare declines under the heaviest dieback scenarios when salvage opportunities are minimal, and when managers make mistakes in regenerating species (for example, by regenerating the wrong species for a number of years before finally finding the species that are best adapted to a new climate).

Differences in how the models treat capital investments within the timber-processing sector have consequences for predictions of regional changes in inventories and harvests. For instance, Sohngen and Mendelsohn (1998) assume that investments in capital and machinery are forward looking and efficient. In both their dieback and regeneration scenarios, markets invest in capital and machinery efficiently as climate changes to take advantage of changing growth rates and replanting opportunities, which allow southern species to be replanted further north than their current range. Harvests consequently shift from region to region in both the short and the long run depending on climate effects. Sohngen and Mendelsohn (1998) thus predict that in the long run harvests increase significantly in northern regions. Despite the salvage opportunities, however, harvests are projected to decline in northern regions in the short run under many of the dieback scenarios.

Alternatively, the TAMM/NAPAP/ATLAS/AREACHANGE model used by Joyce et al. (1995), Mills and Haynes (1995) does not allow for forward-looking investments in capital and machinery. That model assumes that capital and machinery adjust to comparative advantage in timber production in line with relative profit margins across regions. For example, although the ecological results suggest that comparative advantage in tree growing shifts north during climate change, this assumption limits shifts in harvests from south to north. In this model, most capital is predicted to remain in southern regions due to the large remaining timber inventory and harvest opportunities over the 50-year projection period. These estimates have no explicit treatment of salvage opportunities related to climate change, an assumption that results in different harvests for southern species than may be likely if salvage opportunities related to climate change are substantial in the future.

Only one study to date has analyzed global impacts (Perez-Garcia et al., 1997). This is an important limitation in most of the studies above because, as Perez-Garcia et al. (1997) show, climate change may have large impacts in other regions of the world. The authors show that these changes have two effects on markets — changes in regional production levels, and global price effects. The global price effects implied by this research could have important implications for the predictions of adaptation in regional models described above. For instance, if global production rises dramatically and lowers global prices more than the regional models predict, adaptation in North America could be limited by profit considerations. Additional research is needed in this area to understand the potential implications for different regions of the globe.

The differences between these models provide a number of important insights into the breadth and limitations of our knowledge about adaptation in timber markets during climate change. First, although dieback may have significant ecological consequences, Sohngen and Mendelsohn (1998) show that dieback provides powerful incentives for markets to adapt. Second, under the transient climate and ecological scenarios considered, transplanting new species to new regions where climate has changed represents a viable economic adaptation to climate change. Third, Joyce et al. (1995), Mills and Haynes (1995), Burton et al. (1998) show how markets adjust the use of different inputs in the production system when comparative advantage changes.

One limitation in all of the models used to date is that they ignore underlying stochastic processes. Clearly, uncertainty exists with respect to the likelihood of species migration, the rate of increase in timber growth rates, and how humans may intervene in that process. Climate change may cause different distributions in the probability of variables, and the models used so far have generally ignored these possibilities.

New technologies represent another method of adapting to climate change, although most studies have not explicitly addressed technological change and climate change. For example, new adhesives have led to increasing production of new classes of wood panels that have displaced many older products. Alternatively, plastic products have continued to displace traditional uses of wood in end products, such as vinyl siding, plastic decking, and plastic and wood fiber composites. For the most part, new technologies have helped markets become more efficient in their use of timber, but it remains unclear at this point whether and how the pace of technological innovation may be affected by future climate change.

Perhaps the most important limitation in all studies to date is that they present only one-way impacts of a

Table 4
Marginal cost estimates for afforestation programs in selected studies (some estimates reproduced from Adams et al., 1999)

| Study | Discount carbon | Approximate annual flux (million metric t C per year) | Marginal cost (dollar per metric t) |
|---|---|---|---|
| *Global estimate* | | | |
| Sedjo (1989) | No | 2900 | $3.50 |
| *US estimates* | | | |
| Moulton and Richards (1990) | No | 23–45 | $9–11 |
| Adams et al. (1993) | No | 29–56 | $13–19 |
| Parks and Hardie (1995) | No | 44–88 | $21–51 |
| Stavins (1999) (delta states only)[a] | Yes | 7 | <$66 |
| Stavins (1999) (national extrapolation) | Yes | 518 | <$136 |
| Plantinga et al. (1999)[b] | Yes | 0.8–3.7 | $15–90 |
| Adams et al. (1999) | Yes | 16–73 | $5–21 |

[a] Stavins estimates are annual equivalent value of discounted carbon. The discount rate is 5%.
[b] Annual flux estimates are not provided in the Plantinga et al. (1999) study; however, the author provided estimates for use in this table.

changing climate on forested ecosystems and consequently, on economic systems. Clearly, adaptation by humans can affect the ecological response, particularly in human-dominated systems such as the US, and it may have implications for the carbon cycle during climate change (Sohngen et al., 1998). Ecologists are increasingly recognizing that the relative roles of physiological (climate, $CO_2$) changes compared with the potential direct effects of human domination of ecosystems need to be reassessed as a basis for understanding how the carbon cycle will change in the future (Schimel et al., 2000). Truly integrated analysis would incorporate these feedbacks between the ecological system, the economic system, carbon flux, and climate.

While a number of different large-scale, aggregate models — for example, the IMAGE 2.0 model (see Goldewijk et al., 1994) and the IGSM model (Prinn et al., 1999) — are exploring these links, we believe that considerable additional research needs to be accomplished in linking ecological and economic adjustments. One of the benefits of these aggregate approaches is that they provide some indication about the response of future demand growth to climate change, an effect most timber market impact studies have ignored to date. Despite this, the aggregate approaches in the literature thus far have some important limitations for predicting the economic response in specific sectors, such as timber markets. For instance, it is unclear how they handle timber age classes, prices, and adaptation. Although the studies of timber market impacts reviewed above make a variety of assumptions about the ecological and economic adjustment, they all show the important role that inventories and prices play in mediating the speed of the economic response to climate change. We believe that these are important attributes of the timber studies reported above, and that aggregate approaches must evolve to capture them appropriately.

## 3. Using forests to mitigate climate change

A wide range of strategies could be examined to increase the storage of carbon in forests and forest products (Hair et al., 1996; Intergovernmental Panel on Climate Change, 1996; Birdsey et al., 2000). These strategies include reducing carbon emissions from forests by reducing the conversion of forests to farmland and other uses (i.e. reducing deforestation), setting aside existing forests from harvest, and reducing biomass burning. Strategies may also include increasing carbon build-up in forests by converting marginal agricultural land to forests (carbon plantations, forest product plantations, short-rotation woody crops (e.g. Alig et al., 2000) or joint product plantations), and enhancing forest management. Other strategies include substituting wood products for more energy-intensive products (Skog et al., 1996), and planting trees in urban and suburban areas to affect urban climate by shading, reducing wind, and evapotranspiration (Nowak, 1993; McPherson and Rowntree, 1993; US Environmental Protection Agency, 1995). This study considers the three mitigation activities that have been most thoroughly examined in the literature, (1) establishing forests; (2) other land management options; and (3) forest product options.

### 3.1. Establishing forests

Over the years, a number of authors have estimated the costs of establishing plantations in agricultural zones. Several of these studies are presented in Table 4. Historically, researchers investigating the costs of carbon sequestration through afforestation have focused on using land value and planting cost data to estimate the direct costs plus the opportunity costs associated with shifting land from one use (agriculture) to another (forestry). While most afforestation studies use entirely different techniques than those used by the climate change impact studies above, several more recent efforts have used forest sector models for these purposes.

One of the first economic studies of afforestation was Sedjo (1989), who considered a large tree-planting program to offset net carbon emissions. Sedjo predicts that 465 million ha of temperate forest plantations could offset the 2.9 billion metric tonnes (1 billion metric tonnes $= 1 \times 10^{15}$ g) of net carbon emissions from industrial sources each year for 30–50 years. This study, however, considered only planting costs and not land opportunity costs, which would dramatically increase the cost.

Moulton and Richards (1990) use land rental payments from the Conservation Reserve Program in the United States and typical forest establishment costs to estimate the costs of establishing carbon plantations on marginal agricultural land in the United States. They find relatively low costs of establishing forests. Parks and Hardie (1995) use a similar database for land costs, but they construct least cost planting curves based on land costs and land quality. Their results suggest slightly higher costs for carbon sequestration (Table 4).

More recent efforts in the US have relied on cross-sectional data and econometric models to control for the effect of land quality on land opportunity costs. These approaches have a number of benefits for estimating costs. One benefit is that they use disaggregated data (typically county level) on land use proportions and the relative market value of land in different activities. This allows the researchers to directly estimate the current opportunity costs associated with converting additional land from one use to another, i.e. afforesting agricultural land. The econometric approaches used thus far, however, are static, and they do not capture price effects either within a region or across regions. Regional, national, and global price effects may be important if large-scale afforestation projects are undertaken (Alig et al., 1997).

With closer attention to land use proportions and existing land opportunity costs, econometric studies tend to predict higher costs for carbon sequestration than previous estimates. Stavins (1999), using empirical evidence from 36 southern US counties, found that the marginal costs of carbon sequestration in forests are relatively linear for small regional programs up to 7 million metric tonnes (1 million metric tonnes $= 1 \times 10^{12}$ g), but that they increase rapidly as program size increases. Extrapolating these results to the entire US, this study suggests that 500 million metric tonnes of carbon can be sequestered in forests each year for under \$136 per metric ton. One reason why these results may be higher than the previous studies is that Stavins discounts carbon fluxes, and he extrapolates broadly based on a relatively small geographic area. For example, land opportunity costs may be lower in other regions compared with the 36 county regions he investigates.

Plantinga et al. (1999) also develop an econometric model of land use to estimate the costs of carbon sequestration in forests in Maine, Wisconsin, and South Carolina. This study also discounts carbon, but the authors provided estimates of undiscounted, annual sequestration shown in Table 4. One interesting result of this paper is that unlike many previous authors who focus on southern regions for tree planting, this study finds that the lowest cost carbon sequestration is likely to occur in northern regions, namely Wisconsin.

One of the first studies to use a Forest Sector model to assess carbon sequestration costs is Adams et al. (1999). They use the same FASOM model (see Alig et al., 1997, 1998) that was used in the Burton et al. (1998) study of climate change impacts. Unlike the econometric approach, FASOM aggregates land in regions, which allows it to estimate changes in the relative value of land in agriculture and forestry endogenously. Their approach leads to higher cost estimates than earlier studies (with the exception of the econometric models) for several reasons. First, unlike the econometric approaches, land opportunity costs are determined endogenously, that is, they depend on the relative quantity and quality of land in agriculture or forestry. Land, consequently, may shift in both directions. Second, cost estimates in earlier studies were limited to direct government payments to producers for planting, and rent subsidies based on a fixed schedule of agricultural land rental values (Adams et al., 1999). FASOM costs are net changes in consumer, producer, and landowner surplus in both agricultural and forest markets.

Third, they capture dynamic influences on timber markets. For example, if afforested agricultural lands can ultimately be harvested, future forest prices would fall, and agricultural land rents would rise relative to forestland rents. This, in turn, would reduce both the incentive to maintain levels of forest management investment and to retain lands in forest cover rather than shifting them to agriculture (see Sedjo et al., 1995 for a similar discussion). Less intensive management or more forest-to-agriculture land movements would reduce the flux effects of the initial response. Ignoring these reactions, as in previous studies, would lower the apparent cost of the strategy. A third reason is that the FASOM projections account for the storage of carbon in wood products after harvest. Ignoring such opportunities in other studies may inflate costs of carbon storage associated with forestry.

### 3.2. Other land management strategies

Fewer studies have addressed other land management strategies for forest carbon sequestration. Enhanced management may come in several forms. For example, it may result from shifting naturally regenerated stands to managed plantations. The carbon implications of moving from naturally regenerated to managed stands are not entirely clear, however. Harmon et al. (1990) argue that naturally regenerated forests sequester more carbon than plantation type forests in the Pacific Northwest, while Kershaw et al. (1993) suggest that the results depend on a wide range of factors.

Alternatively, enhanced management may include activities to increase biomass in existing managed stands — fertilizing, thinning to increase biomass accumulation, irrigation, spraying pesticides, etc. The only published studies that we are aware of that calculate the costs of sequestering carbon with enhanced management are by Moulton and Richards (1990), and Hoen and Solberg (1994). The former study estimates approximately \$50–60 per metric ton for carbon sequestration in US forests in the South and Pacific Northwest. Hoen and Solberg estimate costs for Norwegian forests of approximately \$80 per metric t through increased management of existing forests.

The cost of reduced biomass burning was investigated by Sohngen and Haynes (1997). They found that reducing forest fires by 2–10% in the United States could increase carbon sequestration by 0.5–0.6 million metric t carbon per year between 2000 and 2040. Their analysis assumed that pre-suppression activities would have to be increased in order to reduce fires, and that fire-fighting activities would have to be increased as well. The resulting costs of carbon sequestration are less than \$32 per metric t for reducing US forest fires. Despite this, the ability of forest and fire managers in the United States to reduce forest fires is, nonetheless, debatable (Sampson and Clark, 1996). Fire management over the last 100 years in the United States has been extremely successful in reducing forest fires. This has lead to an accumulation of material in forests that could be highly susceptible to burning in the future. Increases in the probability of future fires were not considered by Sohngen and Haynes (1997).

### 3.3. Forest product options

Estimates of the pool of carbon in forest products are substantial, although estimates of the cost of increasing these pools in the future have generally not been made. Heath et al. (1996) estimate that the pool of carbon stored in products harvested since 1900 is 3.7 billion metric t carbon, while Skog and Nicholson (1998) suggest that this number is closer to 2.7 billion metric t (1 billion metric tonnes = $1 \times 10^{15}$ g). These studies estimate that net annual accumulation of carbon in wood products and landfills in the early 1990s ranged from 37 million metric t per year (Heath et al., 1996) to 61 million metric t per year (Skog and Nicholson, 1998). The reason that these studies differ is that Skog and Nicholson (1998) predict less accumulation of carbon in products and landfills and more wood burned for energy compared with Heath et al. (1996). Several studies have considered the potential scale of future storage by linking carbon models to harvest projections from the TAMM/ATLAS/NAPAP/AREACHANGE model (this model was also used by Joyce et al., 1995 for climate change impact analysis). Estimates by Plantinga and Birdsey (1993), Skog and Nicholson (1998) suggest that annual storage in products and landfills will grow to approximately 70 million metric tonnes per year by 2040.

As with carbon sequestration programs on the land, the question is whether or not there are options to increase the size of this pool above the baseline estimates. Heath et al. (1996) estimate that increasing the percentage of wood harvest stored in products by 50% would increase carbon storage in products and landfills by 6–7 million metric tonnes per year, and that a decrease of 50% would reduce storage in products and landfills by 2–3 million metric tonnes per year. Skog and Nicholson (1998) provide several suggestions for enhancing the stock of carbon stored in wood products, including (1) increasing the use of lignin containing wood products; (2) increasing product recycling; and (3) increasing product use life. Increasing lignin-containing wood products would amount to increasing the amount of solidwood products in the overall product mix.

An earlier paper by Skog et al. (1996) examined more closely recycling and substituting wood for other products in end uses. They point out that wood as a substitute for engineering products such as steel and concrete has its greatest potential to reduce $CO_2$ emissions by reducing energy consumption associated with the production of the substitute end products. They also describe research suggesting that wood products use eight times less energy than their concrete and steel counterparts on average.

With respect to recycling, approximately 29% of paper is currently recycled by domestic mills, and the USDA Forest Service estimates that this may rise to 45% by the year 2040 (Haynes et al., 1995). Skog et al. (1996) predict that increasing this to 60% by the year 2040 could reduce carbon emissions in the paper industry by 36 million metric tonnes carbon per year, and increase carbon storage in forested ecosystems by 8–14 million metric tonnes carbon per year by lowering harvests in forested ecosystems.

### 3.4. Additional policy considerations for afforestation programs

Considerable evidence in the economic literature suggests that the costs of land use policies are influenced by the methods used to implement the programs (see for example Segerson and Miceli, 1998; Wu et al., 1999). We briefly highlight here several of the differences in implementation assumptions among the models described above. We believe that this represents an area for considerable additional future research to clarify which methods are likely to be most cost-effective for carbon sequestration purposes.

One issue involves whether or not to use fixed land rental payments or to offer landowners payments that reflect land quality for carbon sequestration. Stavins (1999) approaches implementation by assuming fixed land rental payment for carbon sequestration across all acres. That study maintains that fixed payments would be cost-effective because it would be less complicated than developing regional estimates for paying for carbon accumulation. Van Kooten et al. (1995), however, argue for a system of subsidies for carbon accumulation through growth, and taxes for harvest. Although such a scheme would likely be more complicated than across the board payments, making payments based on the quality of land for sequestering carbon may be more cost-effective overall. Other recent studies substantiate these results. For example, Plantinga et al. (1999) suggest that northern regions may be cheaper for carbon sequestration because land opportunity costs are lower there. Adams et al. (1999) also suggest that low-cost opportunities are likely to exist in the north, with more emphasis on the vast US hardwood resource.

For tree planting programs, the length of time over which subsidies are paid to the landowner is an important consideration. The Conservation Reserve Program, for instance, pays landowners for only 10–15 years. It is unclear whether landowners will maintain trees for longer under this type of system. There has been little recent research on this topic, but evidence from analyses of earlier subsidy programs for tree planting indicate very high retention rates across regions, even some 30 years after planting (e.g. Alig et al., 1980). The plantations once established tend to stay in a forest use, even if the originally planted stand is harvested. Although this suggests that cost-share programs can be an effective means of stimulating land use conversion to a long-term forest cover, it highlights a potential area for further research because we do not know the percentage of landowners that would have converted their land to forests with or without the subsidies (Lee et al., 1992).

A related issue that has not been addressed is whether only new forest lands are subsidized, or whether existing lands are subsidized as well. If only new lands are subsidized, it is possible that some landowners will engage in strategic behavior by removing existing forestland for a few years, only to enter the carbon program at a later date. Such behavior is particularly plausible on marginal land where carbon subsidies represent a larger share of land rent. Subsidizing all lands can avoid this problem, but may be more expensive. Understanding the differences between these two strategies would be a useful research project.

It is also important to have a clear understanding of the baseline carbon sequestration, locally, regionally, and nationally. The timber models discussed above suggest that the United States will sequester up to 170 million metric tonnes of carbon per year over the next several decades without climate change and without a carbon sequestration program. The potential success of carbon sequestration programs must be assessed on their ability to increase carbon sequestration above this level. Few afforestation studies discussed above have been clear on their assumptions regarding future afforestation in the absence of carbon sequestration programs.

Leakage in forest carbon sequestration programs was first discussed by Alig et al. (1997). Leakage means that carbon sequestration programs have unintended consequences elsewhere in the local, regional, or national market. For instance, Alig et al. (1997) find that large scale tree planting programs that successfully remove land from agriculture have the effect of causing land to shift from forests to agriculture elsewhere. They found nearly a one to one correspondence between the acres shifting into carbon plantations and the acres shifting from forestry back to agriculture for large programs, suggesting that leakage can be substantial for large programs. The study by Adams et al. (1999) described above does explicitly account for this leakage, but none of the other afforestation studies to our knowledge attempt to measure it.

## 4. Conclusions

This study reviews a number of recent studies investigating the forest sector, climate change impacts, adaptation, and mitigation. Although a number of different ecological and economic modeling assumptions have been made, we believe that the economic impact studies provide a number of important insights into both the potential economic impacts and the potential for adaptation. Results of the mitigation studies are somewhat less clear, although a general result is that estimates of costs have risen over time as economic models have become more sophisticated, and as more issues have been addressed.

Unlike the views suggested in Watson et al. (1998), this review leads us to believe that there is great potential within timber markets in the United States for adaptation because markets will adjust as relative prices among alternative timber supplies respond to climate change. The economic models are uniformly optimistic on the role of prices to mediate the adjustment process in harvests, timberland management, and capital investment. The aggregate ecological changes predicted by the models are relatively slow, which allows markets to adjust in a relatively benign manner at a macro scale, especially with forward-looking economic models such as FASOM and the model used in Sohngen and Mendelsohn (1998). In general, the ecological scenarios used by the modelers project that NPP increases or that the range of the most productive species expands, so that the economic impacts are generally positive (+ \$1.3 to + 7.4 billions per year in 1990 US dollars by the middle of the next century). While more negative ecological scenarios could result in negative economic impacts in the United States, we believe that these results show that prices would, nevertheless, provide a powerful incentive to adapt in order to reduce the scale of the economic impacts.

The economic studies have also investigated more specific impacts of climate change. For instance, a number of ecologists have suggested that biogeographical shifts in species distribution imply widespread dieback among existing forests. Sohngen and Mendelsohn (1998) show that while dieback may have significant ecological consequences (25–46% losses of existing stocks by 2060), it also provides incentives for markets to adapt by transplanting new species to new regions where climate has changed. A related point is that the ecological effects of climate change are likely to adjust comparative advantage in timber production from one region to another. The studies suggest that timber processing capital will shift from one region to another over time as a consequence, but some models project relatively rapid adjustments (Burton et al., 1998; Sohngen and Mendelsohn, 1998), while others project slower adjustments (Joyce et al., 1995; Mills and Haynes, 1995; Perez-Garcia et al., 1997). Additional research on this topic would be useful because it has important implications for projections of regional harvests. For instance, the slow adjustment in capital predicted in the studies by Joyce et al. (1995), Mills and Haynes (1995), Perez-Garcia et al. (1997) does not reduce market welfare in part because timber growth rises in most regions. If stocks of trees were affected more rapidly by climate change, such as with the dieback scenario, comparative advantage among regions for growing trees could be altered relatively rapidly, and the assumption of slow capacity adjustment would have larger effects on projected economic outcomes.

Within the mitigation literature, our general finding is that recent studies have suggested that using forests to mitigate climate change impacts may be more expensive than previously thought. We find that there are a number of reasons why these cost estimates may be higher than previously. First, by estimating parameters that predict how landowners have allocated land to different uses in the past, econometric studies provide a more thorough economic examination of the costs of converting land from agriculture to forestry. Second, econometric studies explicitly account for differences in land quality, and as Plantinga et al. (1999) suggest, costs may vary dramatically from region to region within the United States. Future research and monitoring should include examination of actual landowner behavior, both with respect to how landowners respond to any effects of climate change on their forests, and how they are likely to respond to any incentive programs associated with mitigation strategies.

Second, the FASOM model provides a more holistic estimate of costs — it estimates both land rental costs, and subsequent changes in market welfare. It also measures the potential leakage of forests back to agriculture and the reduction in timberland management. Leakage such as this would occur if afforestation programs increased the inventory of trees and reduced the value of timberland management. Third, the more recent studies have opted to discount future carbon flows. This reduces the total quantity of carbon projected to be sequestered because much of the sequestration occurs in future years. While there are reasons from a benefit–cost standpoint to discount carbon, analysts must determine whether quantities have been discounted before they compare estimates from different studies.

In addition to considering the costs of afforestation programs, we also highlight several studies that have investigated alternative methods for sequestering carbon, such as through product market storage or recycling. While there are not comparative estimates of the costs of sequestration in products, the existing studies suggest that the costs of the options are well worth investigating more thoroughly. For instance, recent estimates of increasing product storage by 50% may provide an additional 6–7 million metric tonnes per year, and increasing recycling to 60% of total products by 2040 (the baseline projection is 45% of products recycled by 2040) would result in up to 50 million metric tonnes of additional storage in ecosystems and reduced emissions in the paper industry. By contrast, the most recent (1999) afforestation studies suggest that the marginal cost of carbon sequestration in forests ranges from \$5–66 per metric ton for a range of annual sequestration from less than 1 million metric tons per year to 73 million metric tonnes per year. Understanding the marginal costs of product storage and recycling options would be a useful research topic to undertake.

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