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ANALYSIS

Sequestration offsets versus direct emission reductions: Consideration of environmental co-effects☆

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A B S T R A C T

Atmospheric greenhouse gas accumulation, and consequential temperature increase, can be remedied by emission reductions or agricultural carbon sequestration. Both options have direct and external environmental effects. This paper examines the magnitudes of existing estimates of corresponding external co-effects and how they could affect optimal combination of emission reductions and agricultural carbon sequestration. We discuss whether government intervention, in the form of taxation/subsidization, is justified for internalizing the externalities considering their relative magnitudes, implementation costs and potential benefits that could be derived from such intervention. We conclude that the existing estimates of external benefits from sequestration and emission reduction do not provide enough support for allocating resources to alter the market mix of carbon sequestration and direct emission reduction strategies.

1. Introduction

Over the last few years the issue of global climate change has become a major environmental research topic. Substantial evidence suggests that global temperatures have risen since the industrial revolution, especially during the latter half of the 20th century (National Research Council, 2000; International Panel on Climate Change (IPCC), 2001). Rising temperatures have been linked to atmospheric concentrations of greenhouse gases. IPCC (2001) argues that control of atmospheric concentrations of greenhouse gases (GHG) through net emissions reduction is needed to mitigate climate change. The major GHG emission source in the United States is Energy consumption and production accounting for 86% of total 6934.6 Tg of emissions on a carbon dioxide equivalent basis in 2002 (EPA, 2004). Therefore, if society recognizes the need to restrict total GHG emissions, it is clear that the energy sector will be subject to policies which will reduce net GHG emissions and consequently will reduce U.S.'s share of GHG concentration in the atmosphere relative to no abatement policy scenario.

One way to address this problem is to reduce emissions by energy sector. However, some economists and environmental scientists have pointed out that agricultural and forestry management practices have the potential to reduce net GHG emissions by removing carbon from the air through photosynthetic processes and storing (sequestering) it in carbon sinks (Adams et al., 1999; IPCC, 2001; Antle and Mooney, 2002; Lal et al., 1998; McCarl and Schneider, 1999, 2000; Plantinga and Wu, 2003). Agricultural and forestry management practices have the potential to reduce the level of atmospheric GHG in two ways: (1) by reducing the amount of emissions

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(minimum till, halting harvesting of timber, prevention of fire) and (2) by sequestering carbon dioxide from the atmosphere by photosynthesis and storing it in soil and biomass (i.e., expanding forested area). In 2002 agricultural activities were estimated to account for emissions of 467.1 Tg (1 Tg carbon = 1 million metric tons carbon) CO_2 equivalent, or 6.7% of total U.S. greenhouse gas emissions, while total net sequestration from land-use, land-use change and forestry was 690.7 Tg CO2 equivalent, which represents an offset of approximately 12% of total U.S. CO_2 emissions (EPA, 2004). Some argue that U.S. cropland can fulfill 30% of the U.S. emissions reduction target as stipulated under Kyoto protocol (Lal et al., 1998; Sandor and Skees, 1999).

In addition to mitigating GHG emission reduction activities generate co-effects, which affect the value of those actions. Specifically, environmental co-effects are generated by various GHG mitigation strategies. Agricultural carbon sequestration can also benefit the environment through reducing erosion and improving wildlife habitat and water quality (McCarl and Schneider, 1999; Plantinga and Wu, 2003). Co-effects also arise when emissions are reduced directly by the energy industry (Burtraw et al., 1999, 2003). For example, an action to reduce CO_2 emissions from power plants would simultaneously reduce emissions of nitrogen oxide compounds (NO_x) and in turn diminish consequent pollution induced health effects (Burtraw, 2000; Burtraw et al., 2003). In fact, evidently there is willingness to pay premium price for mitigation activities which improve local environmental quality (Asuka and Okimura, 2005).

Consideration of co-effects in the GHG mitigation arena has largely been discussed on an opportunistic and sectoral specific basis as in the case of agricultural soil sequestration, afforestation/reforestation, ocean sinks, and control of emissions by energy and transportation sectors (Burtraw et al., 1999; McCarl and Schneider, 1999; Plantinga and Wu, 2003). Here we discuss how environmental co-effects influence socially optimal portfolio of mitigation policies. We also examine existing empirical estimates of co-effects' (Table 1) magnitudes and examine whether policy adjustments are justified to accommodate co-effect considerations. First, we discuss the role of co-effects in emission mitigation portfolios consisting of emission reductions by energy sector and sequestration/emission reduction by agricultural and forestry sector. Next, we review and contrast some of the existing estimates of co-effects of GHG mitigation efforts in energy sector and in agriculture/forestry. We also draw attention to the limitations of numerous assumptions made in arriving at existing co-effect estimates. Finally, we provide recommendations and encourage future empirical research in the area of co-effects of GHG mitigation options.

2. Co-effects and emission mitigating portfolios

To examine how socially optimal portfolio of net GHG emission reduction (mitigation) is affected by co-effects we need an economic framework that portrays portfolio composition. To do this we will use the classic externalities model advanced in Baumol and Oates (1975).

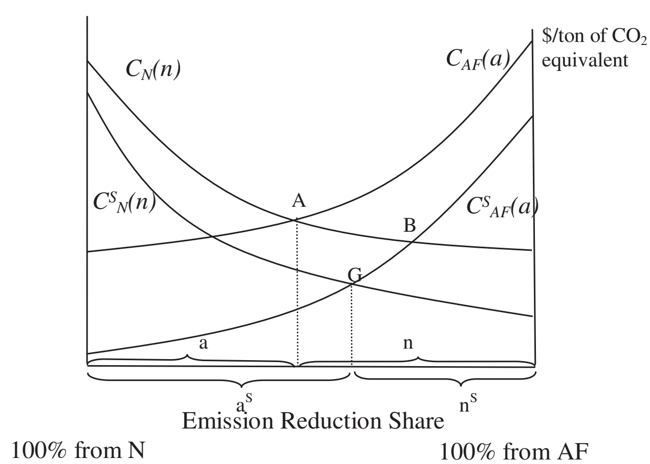
Suppose that the U.S. decides to reduce net emissions to comply with Kyoto protocol and formulates rules that permit a mixed portfolio of sequestration and emission reductions to contribute to this effort. Thus, suppose the energy sector is obligated to satisfy binding emission

Table 1 – Empirical estimates of co-effects

Author(s)	Sector	Activity	Costs	Abatement	Co-effects
Ribaudo (1989)	AF	Afforestation	_	_	\$0.57:\$7.06/ton erosion
Plantinga and Wu (2003)	AF	Afforestation	\$101–132 million/ years	CO_2: 4.1 million tons/years	\$103 million
Pattanayak et al. (in press)	AF	Afforestation,till, crop mix	\$860 million	CO_2: 89.3 million tons/years	2% increase in WQI
Mendelsohn and Balick (1995)	AF	Afforestation	_	_	\$0.9:\$1.3 per ha.
Matthews et al. (2002)	AF	Afforestation	_	_	−1.1%:−2.3% bird population
Pearce and Puroshothaman (1995)	AF	Afforestation	_	_	\$0.01:\$21 per ha/years
Adger et al. (1995)	AF	Afforestation	_	_	\$8/ha
Tobias and Mendelsohn (1991)	AF	Afforestation	_	_	\$52/ha
Walsh et al. (1990)	AF	Afforestation	_	_	\$47/years
Kramer and Mercer (1997)	AF	Afforestation	_	_	\$21:\$31/years
Reaves et al. (1999)	AF	Afforestation	_	_	\$8:\$13/years
Willis and Benson (1989)	AF	Afforestation	_	_	£12:£231 per ha/years
Burtraw et al. (2003)	N	Energy tax	_	CO_2: 40–49 million tons in 2010	\$480:\$980 million in 2010
Mjelde et al. (1984)	N	O_3 emission reduction	_	O_3:–25% _	\$55:\$200 million
Howitt et al. (1984)	N	O_3 emission reduction	_	O_3	\$37 million
Adams and Crocker (1984)	N	O_3 abatement	_	O_3: reduce to 40 ppb	\$2200 million
Kopp et al. (1984)	N	Multiple pollutant	_	Reduce from 120 to 80 ppb	\$1200 million
Felzer et al. (2003)	N	GHG emissions	_	_	\$0.9:\$4.3 trillion
Lee Davis et al. (1997)	N	Fossil fuel	_	_	0.7 million deaths by 2020
Portney and Mullahy (1986)	N	O_3 abatement	_	_	\$0.58 million–0.44 billion

constraints under a tradable emission permits regime. Now suppose the power plants can (a) reduce emissions, including generation of emission allowances and reduction of emissions for the sake of compliance; or (b) purchase emission offsets in the form of carbon sequestration in agriculture/forestry. Emission reductions could arise by lowering the rate of emissions per unit power generated by either improving technological efficiency in the production process or by decreasing production of electricity. Burtraw et al. (1999, 2003) argue that such activities will reduce air pollution in general and thus generate positive co-effects. On the other hand, when sequestration is employed, agricultural external benefits arise but emissions and power plant pollution remain high resulting in high co-effect costs. Therefore, if we rely on agricultural sequestration, net emissions of GHGs may decrease relative to a "do nothing" strategy but net co-effects may or may not decrease relative to "pure emission reductions" due to continued air pollution by power plants.

Emissions offset purchases, whether in the form of Certified Emission Reductions or in the form of emission allowances, occur when the purchaser has a higher marginal cost of abatement than the supplier. Following Baumol and Oates (1975) suppose $C_N(n)$ and $C_{AF}(a)$ are the private marginal GHG emission reduction and sequestration cost functions for the power generation (N) and the agricultural/forestry (AF) sectors respectively. For the sake of simplicity these functions are assumed to be upward sloping and of similar shapes. In Graph 1, the horizontal axis represents the share of total emissions that must be abated by the agricultural/forestry and power generation sectors. Hence, n is a proportion of total targeted emission reductions achieved by the energy sector and a is a proportion of total targeted emission reductions achieved by agriculture and forestry sector. In this notation a corresponds to agriculture and forestry activities that either lead to reduced emissions and/or increased sequestration by agriculture and forestry sector relative to business as usual under no GHG mitigation efforts. Suppose that use of AF strategies generate positive co-effects (Marland et al., 1999; Plantinga and Wu, 2003; Pattanayak et al., 2001). Assuming that the co-effects are quantifiable, social marginal cost function $C^S_{AF}(a)$, which reflects positive co-effects, shifts downward due to decrease of the private costs by the amount equal to co-effect benefits. This function is also reflective of the fact that certified emission reductions generated by agricultural and forestry sector are of temporary nature. Therefore, $C^S_{AF}(a)$ is a social cost function upwardly adjusted by a certain discount factor which accounts for a temporary character of certified emission reductions from AF. Thus, in Graph 1, social AF marginal cost function $C^S_{AF}(a)$ is below the private marginal cost function $C_{AF}(a)$. Also, suppose that reducing emissions from power plants generates positive external benefits. In Graph 1, this can be represented by a shift of the $C_N(n)$ schedule downward to $C^S_N(n)$. If the co-effects of AF differ from those of N, then socially optimal share of emission reductions will differ from the private optimum. In Graph 1, the positive co-effects from sequestration happen to be larger than the positive co-effects from energy sector emission reductions, hence due to the shift, the AF share of GHG mitigation increases while the N share decreases. The key is that whether AF (or conversely N) strategies are preferred from a social standpoint depends on the relative magnitudes of the co-effects from agricultural/forestry sequestration and from energy sector emission reductions.



AF	Agriculture and forestry sector
N	EnergySector
a	Proportion of total emission reductions achieved through AF sector
n	Proportion of total emission reductions achieved through N sector
$C_{AF}(a)$	Marginal private costs of sequestration in AF sector
$C_N(n)$	Marginal private costs of sequestration in N sector
$C^S_{AF}(a)$	Marginal social costs of sequestration in AF sector
$C^S_N(n)$	Marginal social costs of sequestration in N sector

Graph 1 – Implications of externality consideration.

3. Correction of market outcomes

As shown above and developed elsewhere (e.g., Baumol and Oates, 1975), the presence of externalities can lead to market failure and it may be socially desirable for government policy interventions to adjust market outcomes. Theoretically, implementing market subsidies or taxes that reflect net co-effects (NEs) from sequestration or direct emission reduction could correct market failures. However, before such a policy could be implemented, it needs to be discussed whether regulatory intervention in the form of subsidization/taxation is justified based on differences between co-effects from agricultural sequestration and emission reduction by energy sector.

Surely, when comparing agricultural/forestry GHG mitigation efforts to those in energy sector it is important to realize that carbon sequestration by agricultural and forestry sector carries a temporary character, whereas direct emission reduction by energy sector is permanent. However, sequestration in AF allows starting to reduce atmospheric GHG concentrations before cheaper clean energy technologies can be implemented. Keeping this in mind, suppose that marginal cost of sequestration, $C_{AF}(a)$, is implicitly adjusted by permanence discount factor and a factor representing costs savings from shifting abatement from current to future cheaper clean energy technology. Then, minimization of social costs of complying with total emission caps, which correspond to distributed emission permits, would involve consideration of technological costs along with consideration of relative co-effect values of AF sequestration and emission reductions by energy sector. To ensure an interior solution for socially optimal portfolio of mitigation activities it is assumed that it is never optimal for either the AF sector or the power-producing sector to mitigate all emissions[2] (Graph 1). Under such conditions the socially optimal distribution of emission reductions and sequestration levels occur when the following is satisfied.

$$C_{AF}(a) + CE_{AF} = C_N(n) + CE_N \qquad (1)$$

where $C_{AF}(a)$ and $C_N(n)$ are marginal private costs of sequestration and emission reduction for agriculture/forestry *(AF)* and power generation *(N)* activities, while CE_{AF} and CE_N are marginal co-effect values from sequestration and emission reduction, respectively.

Private cost minimization for power-generating industry (*N*) reflects costs of emission reductions as well as payments for purchasing emission permits. Private cost minimization for *AF* would involve technological sequestration costs, opportunity costs, payments received in the form of carbon price, subsidy for sequestering or emitting less than under no GHG mitigation efforts, and market transactions costs associated with CER acquisition. Disregarding transaction costs and again, assuming an interior cost minimizing solution for *AF* participants, which means that it is not privately optimal to sequester to the maximum of their ability because carbon payments do not offset all of the opportunity costs of sequestration, it can be easily shown by using private cost minimization and Eq. (1) that

$$k = CE_{AF} - CE_N = C_N(n) - C_{AF}(a) \qquad (2)$$

where *k* represents the subsidy (or tax) per unit AF offset sold. Hence, in case of interior solution, the optimal level of subsidy would equal the difference of marginal costs and the difference in co-effects simultaneously. This implies that not only do we have to calculate the differences of marginal costs and net marginal co-effects, we also have to figure out at what combination of mitigation levels among agricultural and non-agricultural entities do these differences equal one another (Point G on Graph 1).

There are two points to be made here about computing the tax or subsidy in (2). First, while the above formulation was based on sector level, it is obvious that calculating marginal costs and marginal co-effects on sector level is senseless. Furthermore, even if “average” marginal costs and marginal co-effects where to be calculated, tax/subsidy derived from such estimate will not be effective in terms of internalizing co-effects. It will not be efficient because such value will fail to correctly reflect co-effects generated by individual sequestration and emission reduction efforts. Such value will be too high relative to marginal co-effects of some sequestration projects, and too low relative to marginal values of co-effects generated by other sequestration projects. In other words, co-effect benefits are likely to vary across geographically distant participant regions that use the same strategy to generate emission offsets particularly when it comes to sequestration in agricultural/forestry. For example, reducing tillage intensity in Texas would result in different co-effect benefits as compared to reducing tillage intensity in Nevada. This suggests that the subsidy calculation needs to be carried out on a case by case basis in order to correctly reflect the values of sequestration co-effects. However, the high cost of such a strategy may be unacceptable.

Second, the calculation is complicated by diversity and multiplicity of AF external effects such as improved wildlife habitat, biodiversity implications, improved soil and water quality, development of recreation sites, etc. Each of these external effects is difficult and time-consuming to appraise in terms of dollar values. Evaluation of most of these co-effects requires application of advanced estimation techniques such as non-market valuation analysis, crop production simulation, etc. (Plantinga and Wu, 2003; Ribaudo, 1989; Pattanayak et al., 2001; Matthews et al., 2002). Furthermore, the existence of rental markets for sequestered carbon may complicate the process of evaluating the co-effects. In cases where carbon is rented, appropriate valuation of co-effects would also entail consideration of time horizon over which carbon is stored and co-effects are generated. In addition, sequestration activities in remote regions could result in diverse external effects on biodiversity, soil and water characteristics, among other things (Matthews et al., 2002; Pattanayak et al., 2001), which could be difficult to compare to one another in terms of monetary values. For example, altered mix of bird population,

[2] Possibilities for corner solutions include situations where it is socially optimal for agriculture to sequester to its maximum abilities, and where it is socially and/or privately optimal for power generators to not emit any GHGs. These are unlikely scenarios and are ignored in this analysis.

caused by afforestation, is problematic to appraise relative to prior bird mix.

In order for the subsidization/taxation of sequestration to be economically justifiable the magnitude of the benefits gained from subsidization need to exceed the government expenditures plus transaction costs of estimating the subsidy/tax levels and implementing the corrective policy (McCann and Easter, 2000; Stavins, 1998; Gangadharan, 2000). These costs could be high (Alston and Hurd, 1990).

4. Empirical magnitude of co-effect benefits

Co-effect benefits arise on both the AF sequestration and power plant emission reduction sides. Each has been the subject of some empirical work.

Carbon sequestration in forests or agriculture can generate multiple and diverse co-effect benefits. Agricultural sequestration can improve water quality due to reduced erosion and application of agricultural chemicals and it can alter species habitats. Ribaudo (1989) estimated that water quality damages from erosion and agricultural chemicals ranged from \$0.57 to \$7.06 per ton of eroded soil in the Northern Plains in the Northeast in 1989 dollars. Plantinga and Wu (2003) estimated the reductions in agricultural externalities, which include soil erosion, nitrogen and atrazine pollution, under 5, 10, 15, 20 and 25% conversion rates of Wisconsin agricultural land into forests. They found that under 25% conversion scenario, afforestation would offset 32–42% of the carbon sequestration costs through reduced erosion benefits, where costs of carbon sequestration reflect opportunity costs of agricultural production. Inclusion of soil erosion as well as wildlife habitat benefits generated \$103 million per year, compared to estimated costs of \$101 to \$132 million per year under carbon prices of \$23 and \$30 per ton respectively. Matthews et al. (2002) found that sequestration activities, such as aforestation, affected habitat and in turn biodiversity, showing that converting agricultural land into forests in Southern Carolina, Maine and Southern Wisconsin resulted in altered bird populations. They found that under 10% rate of converting agricultural land into forest land, total bird population decreased by 2.3%, 2.0%, and 1.1% respectively. Pattanayak et al. (in press) found that sequestration programs improved water quality with an improvement in swimmability and aquatic habitat. They analyzed water quality co-benefits form carbon sequestration under \$25 and \$50 per ton of carbon equivalent. It was found that national net agricultural GHG emissions (gross emissions less changes in sequestration and biofuel offsets) declined by about 89.3 million metric tons of carbon equivalent under low price scenario. Under high price scenario agriculture became a net sink of −52.1 million metric ton of carbon equivalent which corresponded to mitigation of 156.3 million metric ton of carbon equivalent per year. Social welfare decreased by 860 million and 3.72 billion under low and high carbon prices respectively due to implications from agricultural production changes. Under both price scenarios average national water quality, measured by water quality index (WQI), which was based on studies by McClelland (1974) and Vaughn (1986), increased by two percentage points.

Expansion of forests could generate benefits in form of pharmaceutical products, ecotourism, endangered species and other non-market values. Walsh et al. (1990) found that, based on use and non-use values, general public was willing to pay as much as \$47/year for forest protection programs in the front range of the Colorado Rocky Mountains. Kramer and Mercer (1997) estimated that U.S. citizens were willing to pay \$21–31/year to create protected areas which would preserve 10% of tropical rain forests. Reaves et al. (1999) found that willingness to pay for restoring South Carolina long-leaf pine forests, which were damaged by a hurricane and provided natural habitat for endangered species, ranged from \$8 to 13/year. The estimates of ecotourism values range from \$8 per ha for Mexican forests (Adger et al., 1995) to \$52 per ha for Monteverde forests in Costa Rica (Tobias and Mendelsohn, 1991). Willis and Benson (1989) examined recreation value of forests in UK and estimated consumer surplus of £1.9 per visit, which corresponded to £12.8–231 per ha. Potential values associated with pharmaceutical products are comparatively lower. For example, Mendelsohn and Balick (1995) suggest that a value of undiscovered tropical forest drugs to the pharmaceutical companies could range from \$0.9 to \$1.3 per ha. Pearce and Puroshothaman (1995) estimate values of \$0.01 to \$21 per ha per year, based on established probabilities of finding a successful drug from plant species in tropical forests.

It has also been argued that carbon sequestration can be viewed as a supplementary device for providing pecuniary co-benefits in terms of economic support to farmers (Marland et al., 1999; Antle, 2000) in the form of possible subsidies for carbon sequestration.

On the energy sector side a decrease in GHG emissions will result in reductions of not only such pollutants as sulphur dioxide (SO_2) and nitrogen oxides (NO_X), but also carbon monoxide (CO), volatile organic compounds (VOCs), particulate matter (PM), and ground-level ozone (O_3) (Chiotti and Urquizo, 1999). Burtraw et al. (1999, 2003) report that reduced CO_2 emissions also lead to reduced NO_x emissions and estimate that this generates health benefits equivalent to \$8.00 per ton of carbon equivalent reduction when carbon emission tax is set at \$25 per ton, corresponding to 30% of the carbon price. Portney and Mullahy (1986) estimated that benefits from avoiding respiratory complications caused by air pollution could range from \$0.58 million to \$0.44 billion. Lee Davis et al. (1997) estimated that by 2020 number of avoidable deaths caused by particulate matter emission from fossil fuel energy would reach 0.7 million under business as usual emission scenario. Chestnutt and Rowe (1989) report results of several studies, which estimate health benefits associated with reducing particulate matter and ozone concentrations. In addition, reduced emission of pollutants also affects wildlife habitat. For example, Scheuhammer (1996) showed that fish in acid impacted environments, which could be created by certain air pollutants, risk increased uptake of certain toxic metals and may accumulate toxic concentrations of these metals more rapidly.

Several studies have shown that by mitigating ground level ozone, which in some cases could be achieved as a by product of reducing GHG emissions, significant benefits could be derived from increased productivity. For example, Mjelde et al. (1984) found that by 25% reduction of ambient O_3 in Illinois, profits of corn, soybean, and wheat producers would increase by $55–200 million. Howitt et al. (1984) found that reduction of ambient O_3 would increase economic surplus by $37 million through California annual crops. Adams and Crocker (1984) estimated that annual national economic surplus would increase by $2200 million if ambient O_3 level was reduced to 40 ppb. Similarly Kopp et al. (1984) estimated that national economic surplus, as affected by corn, soybean, wheat, cotton, and peanuts production, would increase by $1200 million by decreasing federal standards from 120 ppb to 80 ppb. Adams et al. (1986) also show large agricultural benefits from decreased ozone concentrations.

Felzer et al. (2003) argue that damages to the ozone layer of the stratosphere, the incidence of which is affected by emissions from fossil fuel combustion, affect forestry product markets through plant growth, water quantity and quality, nutrient cycling, and recreational opportunities. In addition, Felzer et al. argue that increased ozone damage will reduce terrestrial carbon uptake, which will increase atmospheric GHG concentration and will increase GHG mitigation costs. They estimate that the policy for reducing emissions from fossil fuel combustion would bring about an additional 5–20% of savings in GHG mitigation efforts if we account for the fact that regulation of emissions from fossil fuel combustion would reduce ozone damage and thereby facilitate carbon sequestration. Consequently, GHG mitigation policies will not need to be as stringent to meet the same atmospheric CO_2 goals as in the case when not accounting for additional carbon sequestrations from reduced ozone damage. Burtraw et al. (2003) estimate that inclusion of cost savings emerging from reduced investment in NO_x and SO_2 abatement yields additional benefits bringing total ancillary benefits to about 50% of the $25 per ton carbon tax rate. The Public Services Commission of Wisconsin estimated that the negative co-effects associated with per ton emission of carbon dioxide, methane, and nitrous oxide amounted to $15, $150, and $2700 respectively (EIA, 1995).[3]

It is important to recognize the limitations implied by the broad assumptions that have been made in calculating the values of co-effects estimates mentioned above on both, sequestration and direct emission reduction sides. On agricultural sequestration side Plantinga and Wu's (2003), as well as Ribaudo's (1989), estimates assume perfectly elastic demand for reduced erosion, hunting and non-consumptive uses of wildlife. However, demand is likely to be downward sloping. This would reduce the values of the estimated external benefits derived from AF sequestration. In addition, the majority of studies estimating non-market values of reforestation rely on willingness to pay methodology, which has been criticized in economic literature (Fox et al., 1998; Cummings et al., 1995; List and Shogren, 1998). On direct emission reduction side, Burtraw et al.'s (1999, 2003) estimates do not reflect downward sloping demand for clean air and do not reflect the co-effect costs that could emerge from increased energy scarcity due to decreased energy production. Studies that investigate decreased productivity losses caused by exposure to ambient ozone do not take into account possible behavioral changes which might enable farmers to capitalize on opportunities brought by climate change (Austin et al., 1998). Felzer et al.'s (2003) as well as Lee Davis et al.'s (1997) estimates do not reflect the possibility of optimal trading between various GHG as well as other pollutants. Therefore, they overestimate the costs of GHG mitigation policy.

5. Is co-effects based intervention justified?

Although external benefits arise both from AF sequestration and from power plant emission reductions, the question as to whether policy should be formed to subsidize one type of activity as opposed to the other is certainly in order. The magnitudes of the empirical estimates of the co-effects generated to date may not be so disparate to lead us to the conclusion that the gains from their calculation exceed the costs that might arise due to improper actions stimulated by incomplete estimates plus the transactions costs of generating those estimates. Moreover, given the multitude of co-effects from various GHG mitigation activities, it may prove to be too expensive to calculate values of co-effects from all GHG mitigation activities in order to form policies that would internalize the co-effects from both AF and N activities, and in so doing promote the socially optimal combination of emission mitigation strategies. This may be a situation much like the standard recommendation that secondary benefits not be considered when appraising development projects (Stoevener and Kraynick, 1979; Griffin, 1998). There is a potential welfare gain but the cost and scale of an effort coupled with the inherent uncertainty probably preclude the formation of policy based on such estimates. Therefore, since doing a benefit–cost analysis of any development requires that the project boundaries be very well defined in order to make the problems associated with assigning economic values more manageable, it may be appropriate to exclude co-effects from GHG mitigation efforts from cost–benefit analysis.

6. Summary

This paper discusses the economic balance between the co-benefits of agricultural sequestration and energy sector emission reductions from a social perspective and examines preliminary evidence for implementation. From a social perspective, the optimal portfolio of sequestration and emissions reduction strategies depends not only on costs of abatement but also on net co-effect values. Currently, individual estimates of monetary values of co-effects from sequestration and direct emission reductions both fall in the range of 78% to 50% of the costs of emission control

[3] Corresponding similar values were estimated for Massachusetts and California using marginal costs of pollution control (society's willingness to pay) as proxies to externality values. Society's WTP equals marginal benefit of pollution control, which only equals marginal cost of abatement at the efficient level of pollution. Therefore, the applicability of these estimates is questionable under the context of this paper.

alternatives respectively. Considering the assumptions used in their calculation, these estimates may be close enough to being equal so as to not justify introducing policies to alter the sequestration/emission reduction mix.

Perhaps at this time it is best for case studies to continue to quantify magnitudes of co-effects on regional basis, but for policy implementation to remain focused on GHG emission abatement until compelling evidence of likely beneficial results from subsidization of sequestration process arises. It is also worthwhile informing policy makers about the co-effects involved when emissions are reduced or sequestration promoted. This may help to provide some balance in the face of claims of benefits advanced by some sectoral experts particularly in terms of sequestration activities.

The uncertainties in the magnitudes of co-effects associated with carbon sequestration in agriculture/forestry sector and emission reductions in energy sector call for substantial amount of empirical research in this area. There seems to be a need for development of a methodology for assessing regionally specific and diverse co-effect values from agricultural and forestry carbon sequestration to be compared to the co-effects of direct emission reduction from power generation sector. At the moment the existing estimates of co-effects from carbon sequestration in agriculture/forestry sector and emission reductions in energy sector are based on numerous assumptions, which make it difficult to compare the co-effects for benefit–costs analysis of policies that would alter sequestration/emission reduction mix. Therefore, further research is needed on the evaluation of co-effects from GHG mitigation actions to provide the foundation for a methodology for determining the benefits of including the co-effects into policies related to various GHG mitigation options.

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REFERENCES

Asuka, J., Okimura, T., 2005. Carbon credits: quality and price. Joint Implementation Quarterly. Magazine on the Kyoto Mechanism 11 (3), 8.

Adams, R.M., Crocker, T.D., 1984. Economically relevant response estimation and the value of information: acid deposition. In: Crocker, T.D. (Ed.), Economic Perspectives on Acid Deposition Control. Butterworth Publishers, Boston, pp. 35–64.

Adams, R.M., Hamilton, S.A., McCarl, B.A., 1986. The benefits of pollution control: the case of ozone and U.S. Agriculture. American Journal of Agricultural Economics 68 (4), 886–893.

Adams, D.M., Alig, R.J., McCarl, B.A., Callaway, J.M., Winnet, S.M., 1999. Minimum cost strategies for sequestering carbon in forests. Land Economics 75, 360–374 (Aug).

Adger, N., Brown, K., Cervigni, R., Moran, D., 1995. Total Economic Value of Forests in Mexico. Ambio 24 (5), 286–296.

Alston, J.M., Hurd, B.H., 1990. Some neglected social costs of government spending in farm programs. American Journal of Agricultural Economics 72 (1), 149–156.

Antle, J.M., 2000, Economic Feasibility of Using Carbon Sequestration Policies and Markets to Alleviate Poverty and Enhance Sustainability of the World's Poorest Farmers. Presented at the Expert Workshop on "Carbon Sequestration, Sustainable Agriculture and Poverty Alleviation". World Meteorological Organization, Geneva, Switzerland.

Antle, J.M., Mooney, S., 2002. Designing efficient policies for agricultural soil carbon sequestration. In: Kimble, J. (Ed.), Agriculture Practices and Policies for Carbon Sequestration in Soil. CRC Press LLC, Boca Raton, FL, pp. 323–336.

Austin, D., Krupnick, A., Burtraw, D., Stoessell, T., 1998. The Benefits of Air Pollutant Emissions Reduction in Maryland: results from the Maryland Externalities Screening and Valuation Model. Resources for the Future. http:/www.rff.org.

Baumol, W.G., Oates, W.E., 1975. The Theory of Environmental Policy: Externalities Public Outlays, and the Quality of Life. Prentice-Hall, INC., New Jersey.

Burtraw, D., 2000. Innovation Under the Tradable Sulfur Dioxide Emission Permits Program in the U.S Electricity Sector. Resources for the Future. Discussion Paper No. 00-38.

Burtraw, D., Krupnick, A., Palmer, K., Pul, A., Toman, M., Bloyd, C., 1999. Ancillary Benefits of Reduced Air Pollution in the U.S. from Moderate Greenhouse Gas Mitigation Policies in the Electricity Sector. Resources for the Future. Discussion paper No. 99-51.

Burtraw, D., Krupnick, A., Palmer, K., Paul, A., Toma, M., Bloyd, C., 2003. Ancillary benefits of reduced air pollution in the US from moderate greenhouse gas mitigation policies in the electricity sector. Journal of Environmental Economics and Management 45, 650–673.

Chestnutt, L.G., Rowe, R.D., 1989. Economic measures of the impacts of air pollution on health and visibility. In: MacKenzie, J.J., El-Ashry, M.T. (Eds.), Air Pollution's Toll on Forests and Crops. Yale University Press, Binghampton, NY. Chapter 7.

Chiotti, Q., Urquizo, N., 1999. The Relative Magnitude of the Impacts and Effects of GHG Related Emission Reductions. A Report Prepared for Nevironment Canada. Pollution Probe and Rainmakers Environmental Group. . Available on line at: http://www.msc.ec.gc.ca/saib/atmosphere/docs/co_benefits_e.pdf, last accessed 08/10/05.

Cummings, R.G., Harrison, G.W., Rutstrom, E.E., 1995. Homegrown values and hypothetical surveys: is the dichotomous choice approach incentive compatible? American Economic Review 85, 260–266.

EIA (Energy Information Administration), 1995. Electricity generation and environmental externalities: case studies. Office of Coal Nuclear and Alternative Fuels, Coal and Electric Analysis Branch, U.S. Department of Energy, Washington D.C. 20585.

EPA, 2004. Inventory of U.S. Greenhouse Gas Emissions and Sinks:1990–2002 (http://yosemite.epa.gov/OAR/globalwarming.nsf/content/ResourceCenterPublicationsGHGEmissionsUSEmissionsInventory2004.html).

Felzer, B.S.F., Reilly, J., Melillo, J., Kicklighter, D., Wang, C., Prinn, R., Sarofim, M., Zhuang, Q., 2003. Past and future effects of ozone on net primary production and carbon sequestration using a global biological model. Cambridge, MA, MIT Joint Program on the Science and Policy of Global Change 1, 42.

Fox, J.A., Shogren, J.F., Hayes, D.J., Kliebenstein, J.B., 1998. CVM-X: calibrating contingent values with experimental auction markets. American Journal of Agricultural Economics 80, 455–465.

Gangadharan, L., 2000. Transaction costs in pollution markets: an empirical study. Land Economics 76 (4), 601–614.

Griffin, R.C., 1998. The fundamental principles of cost–benefit analysis. Water Resources Research 34 (8), 2063–2071.

Howitt, R.E., Gossard, T.E., Adams, R.M., 1984. Effects of alternative ozone levels and response data on economic assessments: the

case of California crops. Journal of the Air Pollution Control Association 34, 495–501.
IPCC (International Panel on Climate Change), 2001. Climate Change 2001: The Scientific Basis. IPCC Third Assessment Report. Cambridge University Press, United Kingdom, p. 944 (http://www.ipcc.ch/pub/reports.htm, Accessed 06/16/03).
Kopp, R.J., Vaughan, W.J., Hazilla, M., 1984. Agricultural Sector Benefits Analysis for Ozone: Methods Evaluation and Demonstration. U.S. E.P.A., Office of Air Quality Planning and Standards, Research Triangle Park, NC. Report No. EPA-450/5-84-003.
Kramer, R., Mercer, E., 1997. Valuing a global environmental good: U.S. residents' willingness to pay to protect tropical rain forests. Land Economics 73, 196–210.
Lal, R., Kimble, J.M., Follett, R.F., Cole, C.V., 1998. The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect. Sleeping Bear Press Inc., Chelsea, MI, p. 128.
Lee Davis, D., Kjellstrom, T., Slooff, R., McGartland, A., Atkinson, D., Barbour, W., Hohenstein, W., Nagelhout, P., Woodruff, T., Divita, F., Wislong, J., Deck, L., Schwartz, J., 1997. Short-term improvements in public health from global-climate policies on fossil-fuel combustion: an interim report. The Lancet 350, 1341–1348.
List, J.A., Shogren, J.F, 1998. Calibration of the differences between actual and hypothetical valuations in a field experiment. Journal of Economic Behavior and Organization 37, 193–205.
Marland, G., McCarl, B.A., Schneider, U., 1999. Soil and carbon policy and economics. In: Rosenberg, N.J., Isaurralde, R.C., Malone, E.L. (Eds.), Carbon Sequestration in Soils: Science Monitoring and Beyond. Battelle Press, Columbus, OH, pp. 153–169.
Matthews, S., O'Connor, R., Plantinga, A.J., 2002. Quantifying the impacts on biodiversity of policies for carbon sequestration in forests. Ecological Economics 40 (1), 71–87.
McCann, L., Easter, K.W., 2000. Transaction costs of policies to reduce agricultural phosphorous in the Minnesota river. Land Economics 75 (3), 402–414.
McCarl, B.A., Schneider, U.A., 1999. Curbing greenhouse gases: agriculture's role. Choices 9–12 (First quarter).
McCarl, B.A., Schneider, U.A., 2000. Agriculture's role in a greenhouse gas emission mitigation world: an economic perspective. Review of Agricultural Economics 22 (1), 134–159.
McClelland, N.I., 1974. Water Quality Index Application in the Kansas River Basin. Prepared for U.S. Environmental Protection Agency–Region VII. EPA-907/9-74-001.
Mendelsohn, R., Balick, M., 1995. The value of undiscovered pharmaceuticals in tropical forests. Economic Botany 49 (2), 223–228.
Mjelde, J.W., Adams, R.M., Dixon, B.L., Garcia, P., 1984. Using farmers' actions to measure crop loss due to air pollution. Journal of the Air Pollution Control Association 34, 360–364.
National Research Council, Panel on Reconciling Temperature Observations, Climate Research Committee, Board on Atmospheric Sciences and Climate, Commission on Geosciences, Environment, and Resources, 2000. Reconciling Observation of Global Temperature Change. National Academy Press, Washington, D.C. (http://books.nap.edu/about/availpdf.phtml Accessed 06/16/03).
Pattanayak, S.K., Bondelid, T., Murray, B., Lawrance, D., Yang, J., McCarl, B., Gillig, D., 2001. Water Quality Co-Benefits of Greenhouse Gas Reduction Incentives in Agriculture and Forestry. Report prepared for U.S. Environmental Protection Agency.
Pattanayak, S.K., McCarl, B., Sommer, A., Murray, B., Bondelid, T., Gillig, D., DeAngelo, D.B., 2005. Water quality co-effects of greenhouse gas mitigation in us agriculture. Climatic Change 71 (3), 341–372.
Pearce, D.W., Puroshothaman, S., 1995. The economic value of plant-based pharmaceuticals. In: Swanson, T. (Ed.), Intellectual Property Rights and Biodiversity Conservation. Cambridge University Press, Cambridge, pp. 127–138.
Plantinga, A.J., Wu, J., 2003. Co-benefits from carbon sequestration in forests: evaluating reductions in agricultural externalities from and afforestation policy in Wisconsin. Land Economics 79 (1), 74–85.
Portney, P.R., Mullahy, J., 1986. Urban air quality and acute respiratory illness. Journal of Urban Economics 20, 21–38.
Reaves, D.W., Kramer, R.A., Holmes, T.P., 1999. Does question format matter? Valuing an endangered species. Environ. Res. Econ. 14, 365–383.
Ribaudo, M.O., 1989. Water Quality Benefits from the Conservation Reserve Program. Agricultural Economic Report, vol. 606. U.S. Department of Agriculture, Economic Research Service, Washington, DC.
Sandor, R.L., Skees, J.R., 1999. Creating a market for carbon emissions—opportunities for U.S. farmers. Choices 13–17 (First quarter).
Scheuhammer, A.M., 1996. Influence of reduce dietary calcium on the accumulation and effects of lead, cadmium and aluminium in birds. Environmental Pollution 94, 337–343.
Stavins, R.N., 1998. What can we learn from the grand policy experiment: lessons from SO2 allowance trading. Journal of Economic Perspectives 12 (3), 69–88.
Stoevener, H.H., Kraynick, R.G., 1979. On augmenting community economic performance by new or continuing irrigation developments. American Journal of Agricultural Economics 61 (5), 1115–1123.
Tobias, D., Mendelsohn, R., 1991. Valuing ecotourism in a tropical rain-forest reserve. Ambio 20 (2), 91–93.
Vaughn, W.J., 1986. The Water Quality Ladder. Included as Appendix B in Mitchell, R.C., and R.T. Carson. (1986). The Use of Contingent Valuation Data for Benefit/Cost Analysis inWater Pollution Control. CR-810224-02. Prepared for U.S. Environmental Protection Agency, Office of Policy, Planning, and Evaluation. Washington, DC.
Walsh, R.G., Bjonback, R.D., Aiken, R.A., Rosenthal, D.H., 1990. Estimating the public benefits of protecting forest quality. Journal of Environmental Management 30, 175–189.
Willis, K.G., Benson, J.F., 1989. Recreational values of forests. Forestry 62 (2), 93–110.