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A CONCEPTUAL FRAMEWORK FOR ESTIMATING THE CLIMATE IMPACTS OF LAND-USE CHANGE DUE TO ENERGY CROP PROGRAMS

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Abstract

In this paper I discuss general conceptual issues in the estimation of the impacts of CO₂ emissions from soils and biomass, over time, as a result of land-use change (LUC) due to increased demand for energy crops. The effect of LUC on climate depends generally on the magnitude and timing of changes in soil and plant carbon, and in particular on the timing and extent of the reversion of land to original ecosystems at the end of the bioenergy program. Depending on whether one counts the climate impacts of any reversion of land uses, and how one values future climate-change impacts relative to present impacts, one can estimate anywhere from zero to very large climate impacts due to land-use change (LUC). I argue that the best method is to estimate the net-present-value (NPV) of the impacts of climate change due to LUC. With this approach, one counts the reversion impacts at the end of the program and applies a continuous discounting function to future impacts to express them in present terms. In this case, the impacts of CO₂ emissions from the initial LUC then are at least partially offset by the impacts of CO₂ sequestration from reversion.

Introduction

The establishment and operation of almost any energy production system – not just a biofuel/energy-crop system – changes land use. (If the feedstock is a waste material such as used cooking oil or vented natural gas, then the “production” of the feedstock does not affect land use, aside from trivial second-order effects associated with materials and equipment used in the production process, fuel distribution, and so on.) Surface coal mining, for example, destroys vegetation and disturbs soil over large areas, and the construction of oil pipelines can clear long swaths of vegetation.

However, the land usage of bioenergy crop systems,¹ measured in land area affected per unit of fuel energy produced, is much higher than that of the most land-intensive fossil-fuel production system, surface coal mining. I estimate that surface coal mining produces about 10,000 short tons of coal per acre (based on Howard [1971] and the Energy Information Administration [1996]), which, at 21 million BTU per short ton of coal (Energy Information Administration, 2007) translates into a land requirement of approximately 4,800 acres per quad (1840 hectares per exaJoule). Assuming 5 short tons per acre per year for switchgrass (Walsh et al., 2003), 75 years of production, and 16 millino BTUs per ton of switchgrass (Wright et al., 2006), the switchgrass-energy system

¹ Throughout I may refer interchangeably to “bioenergy” and “biofuels,” but in any case I mean bioenergy derived from energy crops (as opposed to biowastes).

requires 167,000 acres per quad (64,000 hectares per exajoule), or 35 times more land per exajoule than does the coal system.

Petroleum-energy systems undoubtedly occupy much less land per unit of energy than do coal-energy systems, and probably at least two orders of magnitude less land per unit of energy than do bioenergy systems. As a result, it is much more important to estimate the climate impact of land-use changes for bioenergy crop systems than for the major fossil-fuel systems.

In this paper I discuss general conceptual issues in the estimation of the impacts of CO₂ emissions from soils and biomass, over time, as a result of LUC due to increased demand for bioenergy feedstocks. Two issues are especially important: to what extent land uses will revert towards their original conditions when the bioenergy program ends and cultivation is abandoned, and how in general one should treat future climate-change impacts relative to present impacts.

I begin with a discussion of CO₂ emissions from LUC in the larger context of total CO₂-equivalent greenhouse-gas emissions from the lifecycle of bioenergy. I then give a general overview of CO₂ emissions and climate change over time due to LUC, followed by a review of some of the important literature. Next, I focus on the two important issues mentioned above: the reversion of land use to original ecosystems at the end of the bioenergy program, and the treatment of future climate-change impacts relative to present impacts. I point out that depending on whether one counts the climate impacts of any reversion of land uses, and how (or whether) one discounts the impacts of future climate change, one can estimate anywhere from zero to very large climate impacts due to LUC.

CO₂ emissions from LUC in the larger context of CO₂-equivalent greenhouse-gas emissions from the lifecycle of bioenergy

Figure 1 provides a schematic representation of the effects of bioenergy policies on climate and the impacts of climate change. In general, bioenergy policies can directly affect energy, materials, land, and economic systems, which then can affect one-another indirectly through economic (price) linkages. Direct and indirect changes in energy, materials, and land systems can affect climate in a number of ways, primarily by affecting emissions of so-called “greenhouse gases” (GHGs) but also by affecting other climate-relevant characteristics of ecosystems, such as geophysical properties of land.

Changes in land use and the associated changes in climate impacts thus are just one part of a complex web that links bioenergy policies with climate change. Moreover, as indicated by Figure 1, changes in land-use can affect climate in several ways:

- by affecting the flows of carbon between the atmosphere and soil and plants;
- by affecting climate-relevant physical properties of land, such as its albedo (Bala et al, 2007; Pyke and Andelman, 2007; Notaro et al., 2007; Lobell et al., 2006; Feddema et al., 2005; Lamptey et al., 2005; Marland et al., 2003);
- by affecting the nitrogen cycle, which in turn can affect climate in several ways, for example via production of N₂O or by affecting the growth of plants which in turn affects C-CO₂ removal from the atmosphere via photosynthesis (Mosier et al, 2002; Vitousek et al., 1997; Galloway et al., 2003, 2004);
- by affecting the hydrologic cycle, which again affects climate in several ways, for example via the direct radiative forcing of water vapor, via

evapotranspirative cooling, via cloud formation, or via rainfall and thus the growth of and hence carbon sequestration in plants (Bala et al, 2007; Pyke and Andelman, 2007; Lobell et al., 2006; Pielke, 2005; Marland et al., 2003); and

- by affecting the fluxes of other pollutants that can affect climate, such as CH₄, volatile organic compounds, and aerosols.

Generally speaking, the first two factors (C flows, physical properties) have the largest impact on climate, and the last two (hydrodynamics, other pollutants) have the smallest.

Thus, CO₂ emission from plants and soils due to LUC is just one of several ways that LUC can affect climate, and LUC, in turn, is just one of several consequences of bioenergy policies that can affect climate. However, this does not mean that the climate impact of CO₂ emissions from LUC is small; indeed, several analyses have suggested that CO₂ emissions from LUC could be a large fraction of total CO₂-equivalent GHG emissions from the entire lifecycle of biofuels (DeLuchi, 1993b; Delucchi, 2006; Concawe et al., 2006; Righelato and Spracklen, 2007; Reijnders and Huijbregts, 2008s).

CO₂ emissions from LUC are a function of the extent to which bioenergy production induces LUC, the C content of plants and soils in uncultivated relative to cultivated ecosystems, the extent and timing of the reversion of cultivated land when the biofuels program ends, and how one treats future climate-change impacts relative to present impacts. The latter two – reversion of land use and treatment of future impacts relative to present – are a main focus of this paper.

Overview of biofuels, LUC, and changes in C emissions from soils and plants

Any bioenergy program that involves dedicated energy crops, whether corn, soybeans, switchgrass, trees, or something else, inevitably will put pressure on land uses worldwide and bring into cultivation lands that otherwise – in the absence of a biofuels program – would not have been cultivated. (As indicated above, a bioenergy program that uses true waste material – i.e., material that has absolutely no other use – will not give rise to any land-use change.) This effect may be very indirect – for example, corn planting in the U. S. may affect soybean markets in the U. S., which in turn may affect soybean markets in Brazil, which may affect cattle farming in Brazil, which finally may affect deforestation in the Amazon (Laurance, 2007) – but in any event, bioenergy programs ultimately will change land use. Generally, a change in land use entails a change in the characteristics of the soil and the vegetation of the land. In the present context, the relevant characteristic of the soil and the vegetation is its carbon content, which one may express as kg-C per m² of land. Because carbon is exchanged between plants, soils, and the atmosphere, any change in the equilibrium carbon content of plants or soils changes the carbon content of the atmosphere and hence is tantamount to a positive or negative flux of CO₂ to the atmosphere. The impact of this positive or negative CO₂ flux can, after certain adjustments, be added to other GHG emission impacts from the lifecycle of bioenergy to produce a comprehensive measure of the impact of bioenergy on climate.

Conceptually, an ideal model of the climate impact of changes in carbon emissions due to changes in land use caused by bioenergy policies would have several components, listed in Table 1. Table 1 also shows how each component of the ideal model is treated in an actual model developed by the author, the Lifecycle Emissions Model (LEM) (Delucchi, 2003). The LEM estimates CO₂-equivalent emissions of GHGs

from the complete lifecycle of a wide range of transportation fuels, including several biofuels, and has an extensive treatment of emissions from land-use change. Later in this paper I use the LEM to provide illustrative estimates of CO₂ emissions from land use change.

Ideally, emissions of CO₂ from changes in land use would be estimated based on the difference, over time, between ecosystem carbon content in a “no bioenergy program” baseline case compared with ecosystem carbon content in a “with bioenergy program” case, where “bioenergy program” refers to a specific program and need not encompass all bioenergy in the world. To represent this one would create an economic/land-use model with dynamic, price-endogenous, supply and demand functions, with land supply treated explicitly, and with yields determined as a function of endogenous parameters (such as price) and exogenous parameters (such as government R&D policy). One would run this model once, with no bioenergy program, to establish a dynamic “no bioenergy program” land-use baseline (i.e., one in which prices, yields, supply curves, and land uses change year by year), and then run it again for a “with bioenergy program” case, simulated by an outward shift of demand at time zero and then reverse shifts following the end of the program. (The expansion of demand at the beginning of the program and the contraction at the end of the program may be spread out over years, and they may be spread out unevenly, and they may not be symmetrical.) One then would compare land uses between the two cases (“no program” and “with program”) year-by-year, for as long as there remain differences between the two cases (component #1 in Table 1). For each year that there was a difference in land use, one would estimate the change in carbon stocks and emissions (component #2 in Table 1) and then the change in atmospheric CO₂ (component #3 in Table 1), the change in radiative forcing and climate (component #4), and the change in climate impacts (component #5). One would then track these changes in carbon stocks and climate for every land-use category every year. The *impacts* of climate change in each year then would be expressed in the values of a reference year (component #6); in any cost-benefit or economic framework, this would be done by discounting the impacts to their present value. (I discuss this more later.) The sum of the reference-year values of each stream of the impacts of climate change – associated ultimately with the year-by-year differences in land uses between the “no-bioenergy-program” and “with-bioenergy-program” cases – would represent the climate-change impact of CO₂ emissions from land-use changes resulting from a bioenergy program.

Ideally, this modeling would be part of a comprehensive analysis of the climate impacts of bioenergy programs, which would include, in addition to the impacts of CO₂ emissions from LUC just described, two other general kinds of impacts: the climate impacts of LUC *other* than those resulting from CO₂ emissions (e.g., changes in albedo), and the climate impacts from the rest of the bioenergy production-and-use chain (see Figure 1). The value of all of these other impacts would be added to the value of the impacts of the CO₂ emissions from LUC (estimated as described in the preceding paragraph) to produce a comprehensive measure of the climate impact of a bioenergy program.

Note that reality and hence the ideal representation comprise a hierarchy of several separate streams over time: policy streams generate LUC streams, which generate soil-C and plant-C change streams, which generate CO₂ concentration change streams, which generate climate-change streams, which finally generate climate-impact-change streams (Table 2). An accurate representation of the climate impacts of bioenergy program should have an explicit treatment of these streams and a method for

making impact streams with different time profiles commensurate. As indicated in Table 2, the LEM (which I will use later to make illustrative, qualitative estimates of CO₂ emissions from LUC) does represent most of the streams identified in Table 2, albeit relatively crudely in some cases.

With this general background and context, I now review some of the relevant literature on LUC in lifecycle analyses of bioenergy, with particular attention to the treatment of land reversion and the treatment of future climate-change impacts relative to present impacts.

Overview of some of the relevant literature on climate impacts of LUC in the lifecycle of bioenergy

This literature review is organized into six sections:

- estimates of carbon emissions from global land-use change;
- early research on the lifecycle of bioenergy;
- carbon flows in bioenergy and agro-ecosystems;
- studies of land-use change due to biofuel production;
- current research on LUC in the lifecycle of bioenergy; and
- treatment of carbon emissions from biofuels and LUC in an economic or net-present-value framework.

The review shows that there is a long and rich history of research on all aspects the climate impacts of LUC in biofuel LCA *except* as regards the key conceptual issues addressed in this paper: the treatment of future climate-change impacts relative to present impacts, and the treatment of the end of the biofuels program.

Estimates of carbon emissions from global or regional land-use change.

Researchers have known for at least 25 years that changes in land use, and especially conversion of forests, grasslands, and wetlands to agriculture, can result in large carbon emissions to the atmosphere (e.g. Houghton et al., 1983). Since then, analyses of the emissions from global changes in land use (not in the context of lifecycle analysis) have continued apace (Houghton, 1999, 2003, 2005; Houghton and Hackler, 2001, 2002). Regional analyses also are being done: recently, Schulp et al. (2008) projected future carbon sequestration changes in Europe due to land use change.

Early research on the lifecycle of bioenergy. In the U. S., analyses of GHG emissions from the lifecycle of biofuels began in the mid-to-late 1980s, in response to then-newly developing concerns about climate change (see DeLuchi [1991] for a review of some of the early studies). Lifecycle analyses of biofuels were conducted in Europe at least as early as the early 1990s (e.g., Ecotraffic AB, 1992). Some of these early lifecycle analysts recognized that producing bioenergy feedstocks can change land use or cultivation practices and thereby change the carbon content of soil and biomass on the land (e.g. DeLuchi et al., 1987a, 1987b, 1989; Marland and Turhollow, 1990, 1991). In their first lifecycle analyses of GHG emissions from transportation fuels, DeLuchi et al. (1987a,d 1987b, 1989) noted that converting forests to agricultural land to grow crops for biofuels could cause significant reductions in the carbon content of soils and biomass, but they did not provide extensive discussion or quantitative analysis. In their lifecycle analysis of CO₂ emissions from the production and use of ethanol from corn, Marland and Turhollow (1990, 1991) briefly discuss CO₂ emissions from soil: “Long-term corn production usually results in net oxidation of soil carbon, but after some time an equilibrium level is established. The flux is generally small and will not be considered

further” (Marland and Turhollow, 1991, p. 1307). Note that Marland and Turhollow (1990, 1991) refer here to the change in soil carbon due to long-term intensive corn farming on agricultural land, not to the change in soil carbon due to changing the land use from, say, grasslands to corn farming.

In the early 1990s, DeLuchi (1991, 1993a, 1993b) published a comprehensive analysis of the potential magnitude of the change in carbon content of soils and biomass due to biofuel crops replacing forests or grasslands, in the context of lifecycle analysis of transportation fuels. DeLuchi (1991) estimated that clearing a forest to plant corn would result in one-time CO₂ emissions equivalent to 50 to 60 years of emissions from the rest of the corn-ethanol fuelcycle; that clearing grassland to plant corn would result in one-time emissions equivalent to 5 to 6 years of fuelcycle emissions; and that planting short-rotation intensive-cultivation trees over grassland would sequester CO₂ equivalent to about 30 years of emissions from the rest of the wood-to-ethanol fuelcycle. However, DeLuchi’s (1991) early work did not account for the reversion of cultivated land to original ecosystems at the end of the bioenergy program and did not explicitly address weighting or discounting impact streams as a function of time.

Carbon flows in bioenergy and agro-ecosystems. By the the mid-1990s (and perhaps earlier), other analyses had begun to focus independently on the problem of estimating carbon flows in bioenergy systems, albeit not as part of a lifecycle analysis of bioenergy (Marland and Schlamadinger, 1995; Boman and Turnbull, 1997; Schlamadinger et al., 1997; Matthews, 2001). Marland and Schlamadinger (1995) discussed issues involved in estimating the GHG emissions from biofuels, including treatment of by-products, temporal variations in carbon flows between biomass systems and the atmosphere, the impact of changes in land use on soil and biomass carbon, and the possibility of using forests to store carbon rather than produce biofuels:

The possibility of storing carbon in the ecosystem also suggests that if land resources are limited there is an opportunity cost associated with biofuels systems. When land is afforested there will be a net decrease in atmospheric CO₂ emissions whether or not the wood is harvested as a fuel. In fact, [we show] a situation in which, over a considerable time, the opportunity cost in CO₂ is greater than the CO₂ benefit of the biofuels/ wood-products system and the forest is best left standing unless we are prepared to consider a project lifetime greater than 100 years (p. 1135).

Continuing in this vein, Marland and Schlamadinger (1995) pointed out that “biofuels systems require a large resource commitment (land) and a greenhouse-gas assessment should consider the opportunity for using the land in other ways to minimize net greenhouse-gas emissions” (p. 1136). This might mean, for example, restoring land to original native ecosystems rather than developing it for biofuels. Although such alternative land uses might be difficult to implement, and would not always result in lower GHG emissions than would developing the land for biofuels (the balance would depend on the usual details of lifecycle comparisons), the reminder to consider alternative land uses is an important point, one which often has been overlooked in discussions today about the role of bioenergy in mitigating climate change. (As discussed later, a few recent analyses have addressed this point [Righelato and Spracklen, 2007; Piñeiro et al., 2009].)

Boman and Turnbull (1997), referring to papers by Marland and colleagues and others, made broadly similar observations, noting that converting cropland to short-rotation woody crops (SRWC) would increase soil carbon, but that converting forests to SRWC would deplete soil carbon.

Schlamadinger et al. (1997) provided a general discussion of methodological issues in the estimation of the GHG impacts of biomass systems. Similarly, Matthews (2001) developed a “standard methodology” for evaluating the energy and carbon budgets of biofuel production systems, including definition of system boundary, estimation of energy benefits, estimation of carbon sequestration, estimation of energy costs, and estimation of carbon emissions. Matthews (2001) noted that a change of land use from agricultural land to SRWC would increase carbon content, but that a change from forest to SRWC might reduce carbon content, and a change from peat bogs to SRWC could substantially reduce the carbon content of the land.

Several researches have focused specifically on the question of C sequestration in agro-ecosystems (Hakamata et al., 1997; Sauerbeck, 2001; Marland et al., 2001; West and Marland, 2002). Sauerbeck (2001) analyzed CO₂ emissions and C sequestration in agricultural and biofuel systems, and noted (as have many others) that converting forest, grassland, and especially wetlands to agriculture results in large losses of soil carbon. Conversely, Sauerbeck (2001) noted that abandoning surplus farmland and restoring the original wetland or upland soils can increase soil-C contents. Marland et al. (2001) discussed the policy and economic issues associated with the sequestration of carbon in agricultural soils, and West and Marland (2002) analyzed the net carbon flux from agro-ecosystems, accounting for changes in carbon sequestration and emission rates over time. As part of this project, West and Marland (2002) developed a database of long-term experiments regarding soil management.

Studies of land use change due to biofuel production. As indicated in Table 1, the first step in an ideal analysis of C emissions due to land use change is to estimate the relationship between changes in biofuel production and changes in land use. In the U. S., the impact of biofuels on agricultural production was examined formally, with an agricultural/economic model, as early as 1998, when the U. S. Department of Agriculture (USDA), partly in response to the land-use issues raised by DeLuchi (1991, 1993), simulated the effects of increased ethanol demand on planted acreage, crop production, and agricultural exports (cited in Wang, 1999, p. 79). However, because the USDA study did not estimate effects on planted acreage in countries other than the U. S., and did not provide any information on what *kinds* of new land would be brought into production, it could not model how land uses would change globally in response to increased ethanol demand.

Sands and Leimbach (2003) used a partial-equilibrium agriculture and land-use model to simulate global changes in land use in response to carbon policies that create incentives for the production of commercial biomass/biofuel feedstocks, and found that as the amount of land producing biomass/biofuel feedstock increases, the amount of unmanaged land and the amount of managed forest land decreased globally, leading to increased CO₂ emissions from land-use change. Sands and Leimbach (2003) concluded that “policy makers should keep this effect in mind, so that the effect of higher carbon prices does not lead to a biomass-oriented policy that contradicts its own objectives” (p. 205).

Searchinger et al. (2008) were the first to formally model land use change due to biofuel production in the context of a lifecycle analysis of GHG emissions from biofuels. They used the international FAPRI (Food and Agricultural Policy Research Institute) model, a multi-market, partial-equilibrium model of world agriculture, food, fiber, and bioenergy markets (see Fabiosa et al, 2009), to determine increases in cropland worldwide due to the production of biofuels in the U. S. They then assumed that the modeled increases in cultivation encroach into native ecosystems according to the

patterns of conversion of native systems to agriculture observed in the 1990s. Finally, they calculated the carbon emissions associated with the assumed conversion of native ecosystems to agriculture.

Panichelli and Gnansounou (2008) developed a constrained non-linear programming model that allocated biofuel production to different types of land uses subject to assumptions and constraints regarding total demand, yields, total production and trade, and other factors, and then calculated “carbon payback times” given assumptions regarding emissions from land-use change by type of land use. They applied the model to a hypothetical case of soybean-based biodiesel.

Özdemir et al. (2009) point out that models of LUC due to biofuels should account for the effects of the by-products or co-products of biofuel production (e.g. oil cakes of rape, palm and soy are by-products of the production of biodiesel). These byproducts can substitute for other agricultural commodities and hence reduce the land area devoted to agriculture. This reduction in agricultural land area can be a large fraction (e.g., on the order of 1/3rd) of the initial expansion of agricultural land area for growing the biofuel feedstock.

In the past few years there has been an expansion of interest in formal modeling of land use change. Conferences held in Copenhagen, Denmark in 2007 (www.biofuelassessment.dtu.dk/) and Berkeley, California in 2008 (www.edf.org/documents/8883_LCFAC_Workshop_summary.pdf), and a workshop planned for Argonne, Illinois in October 2009 (www.crao.org/workshops/index.html), feature extensive discussions of the issue of the impact of biofuels on land use. Hertel et al. (2009) have edited a volume of contributions on the state of the art in modeling land-use change in the broader context of climate change policies. Darlington’s (2009) gray-literature report provides a review of the main modeling systems and modeling issues.

Current research on LUC in the lifecycle of bioenergy. Recently, researchers have published general methodological discussions of LUC and related issues specifically in the context of lifecycle analysis (Reijnders and Huijbregts, 2003; Koellner and Scholz, 2007; Milà i Canals et al., 2007a, 2007b; Rabl et al., 2007; Kløverpris et al., 2008; see Gnansounou et al. [2008] for an overview of current work). Reijnders and Huijbregts (2003) provided an extensive discussion of LUC in the context of LCA of forest-derived biofuels. They examine land-use prior to biomass growth, the growth of biomass, the fate of the biomass after harvesting, and post-harvesting land use. They noted that “different types of land-use give rise to different levels of carbon in soils,” and that “if there is a change in land-use such as conversion to agricultural land with annual crops, the emission factor will usually increase because the carbon content of agricultural soils is usually substantially lower than that of forest soils” (p. 528). Reijnders and Huijbregts (2003) also cited studies that indicate that, contrary to what is typically assumed, some forests in Europe are *not* in carbon equilibrium, but rather are ongoing net sinks of carbon. Milà i Canals et al. (2007a, 2007b) discuss the time-history of “damage” to land and the issue of what would happen to land in the alternative. Similarly, Koellner and Scholz (2007) develop a framework for representing the damages to the land from LUC, with special attention to the timing and extent of restoration. Importantly, Koellner and Scholz (2007) estimate the “damages” due to land occupation over time, allowing that land restoration can provide benefits (negative damages). Rabl et al. (2007) emphasize that in biofuel LCA “different processes may have very different scales,” and that “it is not appropriate to ignore such delays, even if one does not use monetary valuation and discounting in quantifying the damage costs associated with climate change” (p. 281). Kløverpris et al. (2008) discuss “concepts for modelling how crop consumption affects

the global agricultural area and the intensity of crop production” (p. 13), as part of lifecycle analysis of biofuels. However, none of these papers develop a formal method for evaluating impact streams as a function of when they occur, or consider LUC and reversion in the context of biofuels, or actually estimate the GHG impact of changes in land use due to bioenergy programs.

Today, most discussions of the climate-change and environmental impacts of bioenergy at least mention the importance of LUC, and many recommend that emissions from land use change be included in lifecycle analysis (e.g., *Criteria for Sustainable Biomass Production*, 2006, Appendix 5; UN-Energy, 2006, p. 48; Reijnders, 2006, p. 871; Smeets et al., 2006; Farrell et al., 2006; Heller et al., 2003; Charles et al., 2007; Reijnders and Huijbregts, 2007; The Royal Society, 2008; Holzman, 2008; Koh and Ghazoul, 2008; Scharlemann and Laurance, 2008; Börjesson, 2009; Congressional Budget Office, 2009). Only a handful of analyses of sustainable bioenergy production or bioenergy LCA seem to ignore the issue of changes in land use and carbon storage (e.g., Biotechnology Industry Organization, 2006; Kim and Dale, 2006; Lechón et al., 2009).

Recently, a number of publications have included estimates of emissions from LUC in biofuel lifecycles (Wang, 1999; Delucchi, 2003, 2006; Concauwe et al., 2006; Righelato and Spracklen, 2007; Babcock et al. 2007; Adler et al., 2007; Reijnders and Huijbregts, 2008; Fargione et al., 2008; Searchinger et al., 2008; Gibbs et al, 2008; Panichelli and Gnansounou, 2008; Kim et al., 2009; Piñeiro et al, 2009). People making policies to address global warming have begun to consider emissions from LUC in the lifecycle of biofuels (e.g., Farrell et al., 2007; European Parliament, 2007; Renewable Fuels Agency, 2008; Tollefson, 2009; Environmental Protection Agency, 2009).

Of the recent publications cited above that estimate emissions from land-use change, several are especially noteworthy (Righelato and Spracklen, 2007; Delucchi, 2003, 2006; Adler et al. 2007; Searchinger et al., 2008; Fargione et al., 2008; Gibbs et al, 2008; Panichelli and Gnansounou, 2008; Kim et al., 2009; Piñeiro et al., 2009).² Righelato and Spracklen (2007) examine the issue, raised years ago by Marland and Schlamadinger (1995), as to whether biofuel development is the most sustainable use of the land:

If the prime object of policy on biofuels is mitigation of carbon dioxide-drive global warming, policy-makers may be better advised in the short term (30 years or so) to focus on increasing the efficiency of fossil fuel use, to conserve the existing forests and savannahs, and to restore natural forest and grassland habitats on cropland that is not needed for food. In addition to reducing net carbon dioxide flux to the atmosphere, conversion of large areas of land back to secondary forest provides other environmental services...whereas conversion of large areas of land to biofuel crops may place additional strains on the environment (p. 902).

Piñeiro et al. (2009) examine the same issue and come to a similar conclusion: “based on our comprehensive analysis of 142 soil studies, soil C sequestered by setting aside former agricultural land was greater than the C credits generated by planting corn for ethanol on the same land for 40 years and had equal or greater economic net present value” (p. 277). Earlier, Baral and Guha (2004) had found that sequestering carbon in

² In the other publications (Wang, 1999; Concauwe et al., 2006; Babcock et al, 2007; Reijnders and Huijbregts, 2008) the estimates are fairly simple. In addition, Concauwe et al. (2006) do not include its estimates of LUC emissions in its reported total lifecycle emission results, Wang (1999) uses outdated parameter values, and the analysis of Babcock et al. (2007) is largely illustrative.

standing forests is cheaper than carbon offset from substituting woody-biomass-fired steam electricity for coal-fired steam electricity.

Fargione et al. (2008), Gibbs et al. (2008), Searchinger et al. (2008), Kim et al. (2009) and Piñeiro et al. (2009) provide detailed analyses of emissions from land-use change. Fargione et al. (2008) analyze in detail the C emissions from soil and biomass due to displacement of rainforests, peatlands, savannas, or grasslands by biofuel systems. Their analysis and findings are very similar to, but more detailed than, the early analysis done by DeLuchi (1991,1993). Gibbs et al. (2008) perform an analysis similar to that of Fargione et al. (2008), but use more detailed data bases of crop location and yields and better estimates of soil and plant carbon stocks, and calculate the “carbon payback time” under more scenarios of agricultural productivity and biofuel technology. Kim et al. (2009) use the DAYCENT model to estimate changes in above-ground and below-ground carbon as a result of planting corn on grassland or forest land in North America, under different cropping scenarios. Piñeiro et al. (2009) estimate GHG emissions, including soil C emissions, from corn ethanol and cellulosic ethanol compared with setting land aside to return to native vegetation. They estimate the net present value of the GHG fluxes under these different land-use scenarios, assuming a 3% discount rate and a distribution of CO₂ of CO₂ “prices” based on historical data from the European Climate Exchange. Adler et al. (2007) use the DAYCENT model to estimate changes in soil C after planting bioenergy crops (corn, soybeans, alfalfa, hybrid poplar, reed canarygrass, an switchgrass), assuming that the bioenergy crops are planted on land that had supported conventional tillage cropping for 215 years (p. 677), and find that bioenergy crops increase soil C until a new long-term equilibrium is reached. Finally, as mentioned above, Searchinger et al. (2008) and Panichelli and Gnansounou (2008) take the important additional step of formally modeling land use change.

However, none of these studies have a conceptual framework that explicitly addresses both the reversion of land uses at the end of the biofuels program and the treatment of future climate-change impacts relative to present impacts. (As mentioned above, Piñeiro et al. [2009] do analyze the net present value of GHG fluxes under different land-use conversions.)

Treatment of carbon emissions from biofuels and LUC in an economic or net-present-value framework. In the biofuel LCA literature there have been relatively few treatments of carbon emissions from LUC in an economic framework, in which one estimates the discounted net present value of the impacts of C emissions from LUC. In Delucchi (2003), I estimated the net present value of soil CO₂ and plant CO₂ emissions from LUC, using a time-varying discount rate and with accounting for the reversal of the LUC impacts at the end of the biofuels program. The present value was amortized over the life of the biofuels program. Here, I expand the conceptual underpinnings of that framework. Colleagues at U.C. Berkeley, inspired partly by an unpublished draft of the work presented here, are developing a similar but less detailed framework for analyzing emissions from LUC in biofuel LCA (O’Hare et al., 2009).

Although few lifecycle analyses of LUC and biofuels have been done in a net-present-value or cost-benefit framework, there have been many cost-benefit or cost-effectiveness analyses of a broader set of questions, for example concerning carbon storage and forests (e.g., Plantinga et al., 1999; Joos et al., 1999; Fearnside et al., 2000; Boyland, 2006). In their econometric modeling of land use to estimate the marginal costs of afforestation programs in the U. S., Plantinga et al. (1999) use a “discounting” approach in which carbon flows over the course of an afforestation program are

expressed in present-value terms (using a 5% discount rate). They contrast this with a non-discounting approach in which carbon storage is expressed simply as the mean annual flow over the course of a timber rotation. They argue that the present-value method is preferable because it accounts for the time profile of sequestration and produces a measure that can be used in cost-effectiveness and cost-benefit studies. Boyland (2006) makes the same arguments in an analysis of the economics of using forests to increase carbon storage.

Fearnside et al. (2000) discuss accounting for time and discounting in the context of estimating the global warming impact of land-use change and forestry, although theirs is a framework for policy analysis rather than cost-benefit analysis: they refer to discounting as a “moral” rather than an empirical issue, they do not present a comprehensive damage-cost framework, and they incorporate a time horizon – which as we will see later is used in policy analysis but has no physical or economic basis – in addition to discounting.

Methods of analysis: general considerations

In the remainder of this paper, I extend and revise the NPV framework first documented in Delucchi (2003). I begin here with a discussion of the relationship between changes in land use and changes in CO₂ emissions from soils and biomass (component #2 of the “ideal model” of Table 1). Then, because the ultimate objective is to properly add the climate impact of these land-use-change CO₂ emissions to the climate impacts from the other parts of the lifecycle of biofuels (e.g., emissions from the use of fertilizer or from the conversion of crop feedstock into fuel), to produce a *single* lifecycle CO₂-equivalent metric by which different fuels policies can be compared, I outline a method for converting the actual land-use change C emissions into an annualized stream of impacts commensurable with the annual impacts from the rest of the bioenergy lifecycle (component #6 of the “ideal model” of Table 1). An important part of this annualization method is the treatment of what happens when the bioenergy program ends.

Changes in land use, changes in C-CO₂ emissions, changes in C stocks, and changes in climate. Changes in land use affect the oxidation and formation of carbon (C) in plants and soils. These changes in C oxidation and formation are changes in C-CO₂ fluxes between the atmosphere and the land. A decrease in equilibrium C stocks in soils and plants is an emission of CO₂ to the atmosphere, completely equivalent in its climate effects to burning the same amount of C in a fossil fuel.

To understand clearly how a decrease in the carbon stocks of soils and biomass is tantamount to a one-time emission of CO₂, conceptualize the “original” native ecosystem as being a large carbon deposit, like coal in the ground. Likewise, consider the new biofuel-crop ecosystem in equilibrium as being a carbon deposit. Assume that the size of the biofuel-crop system carbon deposit differs from the size of the original ecosystem carbon deposit by the amount X. Given this, the instantaneous replacement of the original ecosystem with the biofuel system changes the total size of the carbon deposit by X, which is tantamount to emitting X units of carbon at time zero. (Note that this is true only if the bio-C stock is in equilibrium; if it is not, then generally a change in the bio-C stock will *not* be equivalent to burning a fossil fuel.)

In reality, the replacement of the original carbon deposit by the biofuel system carbon deposit does not occur instantaneously at time zero. Rather, the original carbon deposit is reduced over some time, often many years, and the new biofuel system carbon is built up over some period of time.

Now consider the case of bioenergy crops replacing an originally undisturbed native ecosystem such as a forest or grassland. As illustrated in Figure 2, the start of cultivation of a bioenergy crop creates three streams of C emission or sequestration:

- the combustion or decay of the original, native ecosystem plant biomass (represented by the dark-green dashed line in Figure 2);
- a change in the C content of the soil (represented by the brown dashed line in Figure 2; note that it doesn't matter if the carbon change is the oxidation of a particular amount of carbon or instead is the foregone sequestration of the same amount of carbon over the same amount of time; what matters are the difference between the carbon stock in the with-land-use-change scenario versus the without-land-use-change scenario, and the period of time over which this difference is realized); and
- the growth/harvest cycles of the bioenergy crop (represented by the bright green dotted cycles in Figure 2).

The rate of decay of the original plant biomass depends on its fate – whether left to rot or decompose, buried, burned, or converted to relatively long-lived products. Figure 2 shows a relatively long decay period, but if the biomass is burned, the decay period of course will be much shorter. In any case, all of the plant carbon oxidizes eventually, at which point plant-CO₂ emissions are zero. At the same time, the carbon content of the soil changes; in my illustration (Figure 2) it declines, because disturbance and cultivation usually reduces the carbon content of soil (Lal, 2007; Intergovernmental Panel on Climate Change, 2006; Ogle et al., 2003; Del Galdo et al., 2003; Guo and Gifford, 2002).³ Eventually the carbon in the soil reaches a new, lower equilibrium level, at which point soil-C inputs balance soil-C losses and *net* soil-C emissions are zero.

In the meantime the C content of the bioenergy crop *system* (which includes the unburned biofuel itself as well as the crop) is cycling. After the first season of growth, the C content of the crop systems reaches its cyclical steady state, which in Figure 2 is represented as overlapping growth-harvest-oxidation cycles.

At some point, production of bioenergy and cultivation of bioenergy feedstocks will cease, and when it does the entire process will be reversed – at least to some extent, somewhere (I will discuss this more later) – albeit generally at a slower rate, and perhaps not culminating in the same type of ecosystem or the same total ecosystem-C content as existed prior to cultivation. The last of the bioenergy-crop/fuel-system C will oxidize. Somewhere, land that would have been cultivated had bioenergy production continued will not be cultivated, and on that land native plants will begin to re-grow and the carbon in the soil will begin to recover (unless extensive desertification and soil erosion has occurred in the interim). These changes are illustrated in Figure 2, beginning with the line labeled “abandon cultivation.”

³ Agricultural soils have relatively low C content because the rate of C input is relatively low and the “gross” rate of C loss relatively high. The rate of C input is relatively low because C input is related to total ecosystem productivity (annual net primary production of biomass), which is relatively low in agro-ecosystems, and because some or most of the biomass in agro-ecosystems is removed from the site rather than left to return C to the soil. The gross rate of C loss is relatively high because of erosion and soil disturbance. The total long-term *net* soil-C loss is quite variable and can range up to 5 kg-C/m² or more depending on the type of ecosystem displaced, local precipitation, temperature, biological activity, soil type, and other factors.

The re-growth of the native plants and the net recovery of soil C are indicated as a negative emission or sequestration of C. (I refer interchangeably to a “negative emission,” “sequestration,” or “avoided emission” of C.) The process of C fixation (sequestration) by plants removes CO₂ from the atmosphere and hence is the reverse of the process of emitting CO₂ to the atmosphere. The removal (or sequestration) of X grams of CO₂ from the atmosphere, relative to some baseline concentration B, has exactly the same the effect on CO₂ concentration as does eliminating (or avoiding) the emission of X grams of CO₂ in the original baseline B. Put another way, the sequestration of X grams of CO₂ exactly cancels the impact of the simultaneous emission of X grams of CO₂.

In Figure 2, I have indicated that re-growth and recovery are rapid initially but then taper and eventually stop when the plants reach maturity and the soil C reaches a new equilibrium. Note that in Figure 2 the re-growth and recovery periods following abandonment of cultivation are longer than the decay and loss periods initiated by cultivation (which will be typical for many ecosystems), but that the total area under each sequestration curve (representing the total amount of C sequestered) is the same as the total area under the corresponding decay curve (representing the total amount of C emitted), which also will be typical for many ecosystems (Houghton and Hackler, 2001).

Figure 3a shows how these three emission streams affect C stocks in the land, the atmosphere, and the ocean, along with atmospheric temperature change, over time. Here, “land” refers specifically to the land that is directly or indirectly affected by the production of biofuel feedstocks, and “ocean” actually includes terrestrial sinks other than those included in the “land” term. For example, increased levels of atmospheric CO₂ may result in increased growth of some plants, although the effect decreases with increasing CO₂, is dependent on nitrogen availability, and may be offset by increased oxidation of soil carbon (Gerber et al., 2004). (Note that an X% change in the stock of C-CO₂ in the atmosphere results in an X% change in the density and the concentration of CO₂, because the volume of the atmosphere and the stock of the main constituents of the atmosphere, N₂ and O₂, remain constant.) The changes in C stocks over time are estimated quantitatively, assuming exponential decay of emission fluxes and atmospheric C stocks according to eq. 1:

$$C_t = C_0 \cdot e^{-\frac{t}{L}} \quad \text{eq. 1}$$

where:

C_t = carbon level at time t

C_0 = carbon level at time 0.

t = time (years).

L = the e-fold lifetime of the decay (years). The e-fold lifetime is the time it takes to decay 37% of the starting value.

The value of the parameters of eq. 1 that generate the curves in Figure 3a are shown in Table 3.

In Figure 3a the emission and sequestration fluxes for soil C, plant C, and bioenergy crop system C have profiles similar to those in Figure 2, and are denoted with the same graphics. (However, the scale of Figures 3a and 3b is such that the emission profile for the bioenergy crop system, labeled “biofuel crop system C (net emission),” is

too small to be seen.) The clearing of the land and the onset of cultivation generates C emissions from soils and plants, which causes the C stock of the land (soil+plants) to decrease and the C stock (CO₂ concentration) of the atmosphere to increase. However, most CO₂ in the atmosphere starts to be removed immediately (albeit slowly); in the representation for Figure 3a, 82.4% is transferred to the ocean (and other terrestrial sinks) with an e-fold time of 93.5 years, and 17.6% remains in the atmosphere essentially forever (eq. 1 and Table 3).⁴ Thus, most carbon lost from the land is transferred initially to the atmosphere but then gradually from the atmosphere to the ocean (and other terrestrial sinks). This is indicated in Figure 3a by the more gradual change and much later peak of the ocean-C stock compared with the atmosphere-C stock.

The abandonment of cultivation and the start of the recovery of the native ecosystem (in year 50 of Figure 3a) immediately starts to increase the land-C stock and decrease the atmosphere C stock. Shortly after abandonment (at around year 77 in Figure 3a) the atmosphere C stock becomes less than its starting value, and, as sequestration of land-C continues, declines further. When the sequestration of land-C begins to slow, the change in atmospheric C begins to taper and very gradually approaches (from below) its starting value. In the meantime the ocean-C stock has begun to decline, and in the long run all three stocks approach their original levels.

The change in temperature (the red line in Figure 3a) follows the change in the atmospheric C stock, but with a lag that represents the thermal inertia of the system. (The lag here about 50 years, following R. Tol's FUND model as reported in Warren et al. [2006]). In Figure 3a the scale for temperature change is not indicated, because I wish to show only the general shape of the temperature-change curve and the location over time of the peaks and valleys, and these do not depend on the temperature scale. Ideally, Figures 3a and 3b also would show the time trend in the value of the *impacts* of temperature change. However, the relationships between temperature change and impacts are too complex to be estimated and represented in this figure. (For a review of impact functions used in models of damage from climate change, see Warren et al. [2006].)

Figure 3a shows the trends in temperature and C stocks assuming that cultivated land starts to revert to the original ecosystem when cultivation ends. Allowing that there may be debate regarding the timing and extent of land reversion (an issue I take up more below), it is instructive to see the effect of reversion on temperature and C-stock trends. Figure 3b shows the same quantities as in Figure 3a, but with no land reversion and hence no C sequestration in soils and plants after cultivation ends. The trendlines in Figure 3b are identical to those of Figure 3a up to the point at which cultivation ends (year 50), but diverge dramatically after that. After a brief emissions bump representing the oxidation of the C from the final production and use cycle of the bioenergy system, the atmosphere C stock and hence the temperature begin to decline gradually – much more gradually than in the “reversion” case, because here the atmosphere C stock is declining *only* because of the gradual transfer of C to the ocean (and other terrestrial sinks), and *not* also because of the re-sequestration of C due to reversion. Moreover, because in this case there is no sequestration or negative emission

⁴ This is a simplification of the actual CO₂ decay functions used in the LEM, which are based on Shine et al. (2005), Archer (2005), and other sources. Shine et al. (2005) assume that 17.6% of anthropogenic CO₂ has an infinite lifetime, but Archer (2005) points out that this fraction has a very long but not infinite lifetime (tens or even hundreds of thousands of years).

of CO₂, the atmospheric C stock and the temperature trend never drop below their starting values, and the ocean C stock does not approach its original value. In the case without reversion, the net result is a transfer of C from the land to the atmosphere, and eventually, to the ocean, with considerably more total degree-years of warming than in the case with land reversion. (One degree-year of warming is a warming of one degree for one year.)

Indeed, under certain conditions, the reversion of land uses can negate entirely all of the degree-years of warming due to the emissions from LUC at the start of the program. These conditions are: that all of the CO₂ emitted due to the initial LUC eventually is re-sequestered by the reversion of land uses; that no anthropogenic CO₂ has a truly infinite lifetime; and that relationships between concentration, radiative forcing, and temperature change remain the same over time. (If any CO₂ emission has an infinite lifetime, then the degree-years of warming that occur from time zero up until reversion cannot be canceled by the degree-years of cooling from a negative emission that starts after reversion.) Of course, in general these conditions do not hold (and in my own lifecycle modeling I do not assume that they hold). Moreover, as I discuss more later, society does not care about degree-years of warming per se, but rather about the value of the impacts of climate change, and for two reasons the present value of the foregone impacts of climate change due to reversion generally will not be the negative of the present value of the impacts of climate change due to the initial LUC.

First, the present value of the impacts of climate change decline with time, so that in general – i.e., all else equal – the climate impacts of reversion will have a lower absolute present value than do the climate impacts of initial LUC. Second, the relationships between temperature change, climate impacts, and the value of climate impacts are complex and will vary over time. For example, it is possible that the climate impacts due to the initial LUC will not include possibly high-cost events such as the collapse of Antarctic ice sheets, but that the sequestration due to the later reversion will avoid those high-cost events; in this case, the climate impacts of land-use reversion may have a higher absolute present value than do the climate impacts of the initial LUC.

Nevertheless, it is clear that reversion has the potential to negate a substantial portion of the initial (and generally large) *value* of the impacts of emissions from initial LUC. (Again, note that I do *not* claim that reversion will return the planet to its original, pre-LUC, *physical* state; rather, I claim that the value of the further damages to the planet foregone may be very close to the value of the damages incurred due to the initial LUC.) Hence, how one treats the extent and the timing of land-use reversion matters a great deal in lifecycle analyses of biofuels.

Note, too, that the time profile of the change in atmospheric CO₂, and hence the time profile of the change in climate and its impacts, would be different if all of the land emissions fluxes and hence all of the change in the land-C stock happened instantaneously: in this case, atmospheric CO₂ would spike at year 1 and then start to decline immediately, and the bulk of the degree-years of warming, and of the impacts of warming, would occur sooner than in Figures 3a and 3b. Again, given that society is not indifferent between the impacts of *H* degree-years of warming today and the impacts of *H* degree-years of warming, say, 85 years from now, the valuation of the impacts of CO₂ emissions – the ultimate quantity of interest – depends on the time-profile of emissions and impacts and the valuation of emission impacts as a function of time.

Reversion of land uses, and discounting future impacts relative to present: overview

For my purposes there are four general approaches to estimating CO₂-emission impacts due to land-use change, depending on how one values future impacts relative to present impacts (discount all future impacts, or don't discount anything up to some time horizon *TH*), and whether one includes the impacts of the sequestration that occurs after the biofuels program and associated cultivation end. The four approaches – discount or do not discount future impacts versus count or do not count sequestration impacts after cultivation ends – are presented in Table 4.

Generally, approach #2 in Tables 4 and 5 – ignore end-of-cultivation (EOC) carbon sequestration and use a non-zero discount rate to amortize the initial emissions due to LUC at the start of the program – will yield the highest net emissions from LUC (Table 5). Approach #4 – ignore EOC sequestration and use a zero discount rate up to threshold time horizon *TH* (implicit or explicit in several studies – see Table 4) – will estimate the next largest LUC emissions, followed by approach #1 (the approach recommended here). As indicated qualitatively in Table 5, approach #3 typically will result in low to zero impacts. Because of this wide range of possible results, it is important to better understand the validity of the different approaches. In order to decide this, I discuss whether it is better to evaluate future climate-change impacts relative to present with a continuous discounting function or with no discounting and a threshold time horizon, and whether one should count sequestration after cultivation.

The treatment of future climate-change impacts relative to present impacts

The issue of how to treat future climate-change impacts relative to present impacts comes up because lifecycle analysts want to combine the impacts of emissions from land-use change (LUC) with the impact of emissions from the rest of the biofuel lifecycle (RBL), but they know that the temporal pattern of emissions from LUC is different from the temporal pattern of emissions from the RBL (LUC emissions tend to occur at the beginning of the program, and RBL emissions tend to occur in a steady stream throughout the life of the program), and they recognize that this difference might affect how they combine LUC impacts with RBL impacts.

Analysts wish to combine LUC impacts with RBL impacts because they wish to have a *single* metric of the overall climate impact of biofuels, such as CO₂-equivalent emissions per KJ of fuel produced. Without such an overall metric, analysts cannot say anything about the overall climate impact of biofuels.

It is of course possible to ignore differences in the timing of LUC vs. RBL emissions and simply divide total grams of emissions from land-use change over some period of time by the total associated kJ of fuel output (whether over the same period of time or not), and add the resultant g/KJ emission factor to the g/KJ emission factor calculated for the rest of the biofuel lifecycle. And this in fact is what most analysts have done (approach #4, Table 4). What is wrong with this approach?

The key to answering this question is to recognize that society cares not about GHG emissions or radiative forcing or even temperature change *per se*, but rather about the *impacts* of climate change caused by GHG emissions. These include impacts on human health, ecosystems, agriculture, tourism, water supply, infrastructure, and so on. Society is concerned about climate change, and is interested in evaluating alternatives with respect to their effect on climate change, to the extent that society is not indifferent about these impacts. It matters not exactly how society goes about evaluating impacts and coming up with some overall assessment; the point – perhaps a tautology – is that society cares about these *impacts* and not about emissions *per se*.

Once this premise is accepted, it is easy to show that the approach of ignoring differences in the timing of LUC vs. RBL emissions and simply dividing emissions by fuel output is invalid. That approach would be valid if and only if the ratio:

$$\frac{\text{emissions from land-use change}}{\text{emissions from the rest of the lifecycle}}$$

were equal to the ratio we care about:

$$\frac{\text{the value of the impacts of emissions from land-use change}}{\text{the value of the impacts of emissions from the rest of the lifecycle}}$$

If the emissions ratio is not equal to the value ratio, then adding LUC emissions to RBL emissions will not give the same answer as will performing the addition on the basis of the value of impacts, and because adding on the basis of the value of the impacts gives the right answer, adding emissions would give the wrong answer.

So, is the emissions ratio equal to the value ratio? The two ratios are equal only if the time pattern of LUC emissions is the same as the pattern of RBL emissions, *or* if society doesn't care about differences in the timing of the impacts of climate-change emissions. It is indisputable that neither condition is true.

Therefore, one cannot validly (or meaningfully) ignore time and divide LUC emissions by fuel output and then add the result to g/kJ emissions from the RBL. Instead, one must follow the procedure outlined in Table 1. This means that one must account for differences in the timing of the impacts of LUC emissions relative to the timing of the impacts of RBL emissions. More formally, we must decide how to evaluate the impacts of a one-unit emission today relative to the impacts of a one-unit emission at some time in the future.

This problem – how to evaluate the impacts of a one-unit emission today relative to the impacts of a one-unit emission at some time in the future – has been discussed widely in the context of developing CO₂-equivalency factors (CEFs) for different greenhouse gases (Fuglestvedt et al., 2003; Godal, 2003; O'Neill, 2003; Manne and Richels, 2001; Bradford, 2001). I will use that discussion as a starting point here. A CEF for gas *g* is the mass amount of *g* that has the same “climate impact,” however defined, as does one gram of CO₂ emitted at the same time. In this CEF context, the problem of estimating impacts and valuation of impacts over time arises because different gases have different lifetimes in the atmosphere and hence have different impact-profiles over time. In the development of CEFs, analysts have employed two methods to address changes in impacts and the value of impacts over time – the same two methods we consider here:

- A) do not discount any impacts up to some time horizon *TH*, and ignore impacts after *TH*; or
- B) discount all future impacts at some rate *r*, and estimate the present value of the impacts of climate change due to emissions, according to the formula:

$$PV = FV \cdot e^{-r \cdot t} \text{ or } PV = \frac{FV}{(1+r)^t} \quad \text{eq. 2}$$

where:

PV = the present (year 0) value of the future (year t) impacts
FV = the future (year t) value of the future (year t) impacts
 r = the discount rate (discussed below)
 t = time

Method A (time horizon) is embodied in approaches 3 and 4 of Tables 4 and 5, and Method B (discounting) is embodied in approaches 1 and 2.

Using a time horizon TH before which everything counts equally and after which nothing counts (method A) is equivalent to having a discount rate $r = 0$ up to $t = TH$ and a discount rate $r = \infty$ for $t > TH$. The problem is that this discontinuous representation is unrealistic. The main forces that govern social valuation as a function of time – uncertainty and risk, changes in underlying valuation parameters, and changes in wealth (Stern and Taylor, 2007; Sherwood, 2007; Bürgenmeier et al., 2006; Goulder and Stavins, 2002; Fehr, 2002; Rabl, 1996) – are likely to be continuous rather than discontinuous functions of time. More importantly, the behavioral assumption implicit in method A – zero discounting in the near term, extremely high discounting of the long term – is the reverse of how people and societies actually seem to behave, which is to discount the future at a lower rate than the present (Cropper et al., 1994; Rabin, 1998; Philibert, 1999; Gollier, 2002; Oxera, 2002; Frederick et al., 2002; Karp, 2005). Thus, whatever one believes about whether and how people compare future impacts with current impacts, method A is not consistent with physical, economic, social, or psychological reality.

Discussing this issue in the context of estimating CEFs for different gases, where many analysts use method A to estimate a quantity called a “Global Warming Potential,” or GWP, Bradford (2001) writes that the use of a time horizon is “clearly wrong:”

To say that emission of an incremental tonne of one gas has the same implications for policy as emission of X tonnes of another gas means that we have to assign a value to the change in radiative forcing at different times in the future. There is no way to avoid this step. Using the physical GWP involves an implicit evaluation: a bit of extra radiative forcing at any time up to the chosen horizon has the same (negative) value as the same bit at any other time within that span; an extra bit beyond the horizon has zero value. This is clearly wrong (Bradford, 2001, p. 650).

Similarly, Tol (2006) asserts that “it is now widely acknowledged that GWPs have limited validity in the natural sciences...and no validity in economics or policy...In a cost-benefit analysis, the proper equivalence...is the ratio of the marginal damage costs” (p. 243).

I agree with Bradford (2001) and Tol (2006). One must make *some* decision about evaluating impacts over time, and even though it is difficult to specify a continuous discounting model to everyone’s satisfaction, that is no reason to adopt a convention – the use of a time horizon – that surely is unrealistic and therefore almost certain to be wrong. The more reasonable approach is to model the physical and economic forces actually at work over time and compare all emission streams on the basis of the present value of their impacts. This will involve the use of continuous discounting. In addition, it is important to emphasize Goulder and Stavins’s (2002) point that “applying a discount rate does not mean giving less weight to the welfare of future generations. Rather, the process simply converts the (full) values of the impacts that occur at different points of time into common units” (p. 674).

Thus, looking at the four approaches to dealing with the issues of time and end-of-cultivation, in Tables 4 and 5, the two approaches that involve a time horizon – approaches 3 and 4 – are not valid.

Focus on the discount rate r . The role of the discount rate r here (eq. 2) perhaps is best understood in the context of cost-benefit analysis of mitigation of climate change (see Bürgenmeier et al., 2006). Any policy question regarding mitigation of climate change can be evaluated by comparing the costs of present mitigation actions with the future climate benefits of mitigation. To express the future benefits (the avoided future impacts of climate change) in present terms, we discount the future benefits at a rate $r = r_c - r_b$, using eq. 2, where r_c is the rate of change in the opportunity costs of investments in climate-change mitigation and r_b is the rate of change in the value of benefits (the avoided impacts of climate change). The discount rate $r_c - r_b$ captures all of the reasons that the value of the impacts of greenhouse gases depend on the time period over which they occur.

This result is similar to the positions of Lind (1995) and Rabl (1996). Rabl (1996) states that “only the growth component of the discount rate [r_c] is relevant for a cost-benefit analysis from the point of view of future generations” (p. 138; I’ve inserted the term in brackets), but he also explicitly accounts for changes in the value of the avoided impacts of climate change (r_b). Lind (1995), talking about the discount rate in the role of cost-benefit analysis of climate change, states that “the fundamental question...hinges on the rate of per-capita income growth [r_c] and on the magnitude of the effect of global warming [r_b]” (p. 380; I’ve inserted the terms in brackets).

Thus, in a cost-benefit analysis of mitigation of climate change, one discounts the dollar value of the avoided future impacts of climate change (i.e., the future benefits) using a rate that represents the difference between the rate of return on alternative investments (r_c) and the rate of change in the value of benefits (r_b). This difference will be relatively close to zero, because the rate of real economic growth (a good estimate of r_c) is relatively low to begin with (probably less than 3%/year globally), and the rate of change in benefits will be subtracted from this. *The discount rate so derived should be applied to the estimated impacts of emissions due to land-use change.*

Philosophical and practical objections to the use of a discount rate. I do recognize, however, that there are significant philosophical and practical objections to the use of cost-benefit analysis and to the estimation and application of a discount rate in the context discussed here. Because climate change affects so many aspects of life on planet earth – in many unpredictable ways, in a wide range of regions, over a very long period of time – it is virtually impossible to model with any reasonable degree of certainty, and hence one may question on conceptual, methodological, and even philosophical grounds the application of cost-benefit analysis to such an analytically intractable problem. One may decide that utilitarianism, which is the philosophical foundation of cost-benefit analysis, is a morally impoverished or otherwise inappropriate framework (Lugwig, 2000; Sagoff, 1988; Ott, 2003), or that available non-market valuation methods applied to large-scale environmental damages from climate change do not produce meaningful results (Neumayer, 1999, 2007; Hanley, 1992; Stirling, 1997),⁵ or that cost-benefit analysis is not helpful because uncertainty in model

⁵ Neumayer (1999) asserts that “the real question is, again, whether large-scale damage to natural capital caused by global warming can be compensated for by higher consumption levels or not” (p. 40). Since the cost-benefit approach values all environmental damages and all consumption opportunity costs in dollars, and a dollar is a dollar irrespective of how it was produced, Neumayer (1999) really is asking

structure and parameter values is so great one can come up with almost any answer one wishes (Azar and Lindgren, 2003).

A specific concern of some researchers is that the possibility of thresholds or “tipping points” in climate change – relatively rapid, large scale, irreversible effects such as the melting of large ice sheets (see Hansen et al., 2007b, 2008) – can confound the use a discount rate as outlined here (e.g., Fearnside et al. [2000] p. 241). In Appendix A, I discuss this issue in some detail, and conclude that the possibility of tipping points, irreversibilities, and “catastrophes” in climate change does not invalidate the estimation of a discount rate, within the context of cost-benefit analysis, for the purpose of equilibrating impacts that occur at different times. The widely cited Stern (2006) report comes to a similar conclusion: “taken together with our discussion of ethics we see that the standard welfare framework is highly relevant as a theoretical basis for assessing strategies and projects in the context of climate change” (p. 52). Likewise, in his discussion of the uncertainties of climate change, Schelling (2007) advises us to “weigh the costs, the benefits, and the probabilities as best as all three are known, and don’t be obsessed with either extreme tail of the distribution.”

Nevertheless, I acknowledge again that there can be legitimate objections to economic discounting in the context considered here. A reasonable alternative course might be to follow what seems to be Weitzman’s (2007, 2008) suggestion and distinguish the more run-of-the-mill outcomes of climate change from the very unlikely but essentially cataclysmic outcomes, and address the former but not the latter within the standard framework of cost-benefit analysis.

Counting the impacts of C sequestration at the end of cultivation (EOC)

I turn now to the question of whether to count the impacts of C sequestration at the end of cultivation (apropos Tables 4 and 5). There are two basic arguments for *not* counting EOC sequestration: a conceptual argument that EOC impacts are not related to the original biofuels development activity whose climate impact is being analyzed, and a theoretical/empirical argument that in any case EOC sequestration is not likely to be large. I examine both these arguments next.

Put more formally, the conceptual argument is that the consequences associated with the end of a bioenergy program are determined not by the original bioenergy policies or market conditions (the ones that spurred the initial development and use of bioenergy, and whose climate impacts are being analyzed), but rather by separate “end-of-life” policies or market conditions. If this view were valid, then EOC sequestration properly would be modeled as a specific consequence of those end-of-life conditions rather than as consequences of the “original” conditions. However, while it is true that the particulars of the end of a bio-energy program may be affected by policies or actions that arguably should be subject to a separate policy analysis (for example, one may want to analyze the impacts of local land-use policies that affect the timing and extent of land reversion in particular regions), any initial biofuels program has two general end-of-life consequences that affect climate and hence must be considered as part of the climate impacts of the original biofuels program. First, the biofuels program will in fact end. Second, the ending will be the general *reversal* of the expansion of demand for biomass feedstocks that marked the start of the program.

whether the non-market valuation methods applied to large-scale environmental damages from climate change produce meaningful results. See also Neumayer (2007).

1. *Biofuels programs will end.* Not only can no biofuels program last literally forever (because nothing can), it is likely that no biofuels program will persist to any significant extent beyond 100 or at the outside 200 years, if only because more sustainable energy options probably will be available within the next 100 years. Certainly, biofuels from agricultural crops, such as corn and soybeans, will be transitional and minor energy sources, at most, and hence will not last more than a few decades. And given that climate change operates over time scales of centuries, and that it is not reasonable to ignore climate impacts even 200 or 300 years from now – because, given the very low discount rates that are likely to obtain in the analysis of the impacts of climate change (see the discussion above), impacts that occur even 300 years from now will not necessarily have a near-zero present value – there are not strong grounds for deciding that the end of a bioenergy program will be too far in the future to be of any concern.

Of course, it is not unreasonable to consider a scenario in which certain kinds of biofuels last so long – say, at least 1,000 years – that the impacts of the reversion at the end have very little present value. But the point here is that it is clearly unreasonable to assert that this is the *only* relevant scenario and that therefore reversion always will be unimportant.

2. *The end of the biofuels program will be the general reversal of the initial expansion of demand for biomass feedstocks.* Regardless of the specific policies or market conditions at the end of the biofuels program, if the start of the program is defined by an expansion of demand for biomass feedstocks, then the end of the program is defined by a contraction of demand for biomass feedstocks. That is, contraction of demand is the *defining* characteristic of the end of the biofuels program: if demand does not contract, then program has not ended. And although this may be obvious, it has an important implication for the analysis of the climate impacts of biofuels policies: when demand contracts, the land-use changes and associated climate impacts wrought by the initial expansion of demand will tend to be reversed.⁶

I can now combine these two consequences into an important single statement: The demand expansion that occurs at the start of a biofuels program eventually will be reversed, probably within a time frame that society care about, and then the land-use changes and climate impacts associated with the initial expansion also will *tend* to be reversed. Note, though, that I say “tend to be reversed,” because there is no theoretical reason that the end of the program must *exactly* reverse the land-use changes and climate impacts of the start of the program. This is because between the start and the end of the biofuels program, changes in population, resources, technology, tastes, and so on can change supply and demand relationships, and given that the land-use impact of a demand shift depends on the shape of the land-supply curve and on *where* along the long-run land-supply curve the shift occurs, changes in the relative positions of

⁶ Some of my colleagues have suggested that even if demand for biofuels does contract, land will not be taken out of production because farmers will want to keep the land in production regardless of the economics (i.e., even in the face of slackening demand and falling prices). I believe that this is incorrect and also partly misses the point. It is incorrect because it is widely accepted by agricultural economists that land goes in *and out* of production depending on agricultural commodity prices and other factors (e.g., www.econ.iastate.edu/faculty/harl/Senate_Testimony_Sept_15.html). It misses the point because the effect of the end of the biofuels program may well be a reduction in the rate of increase of agricultural expansion rather than an actual decrease in land area under cultivation, and in the former case no land actually goes out of production.

demand and supply curves over time can result in different impacts on land use due to a given absolute shift in demand.

This general phenomenon is illustrated in Figure 4, which shows how shifts in demand at the beginning and the end of a biofuels program drive changes in the quantity of land cultivated. At the start of the biofuels program, the total demand curve shifts outward from D_0 to D_1 (an expansion of ΔD due to biofuels), price rises to P_1 , and quantity of land cultivated increases by ΔQ . Now, suppose that over the life of the biofuels program total demand continues to expand, because of growing populations and growing demand for land-intensive forms of food, such as meat. Suppose also that the land-supply curve has the shape shown in Figure 4, in which some maximum amount of cultivatable land Q_{MAX} is approached asymptotically (e.g., van Meijl et al., 2006). In this case, by the end of the biofuels program the demand curve will be at D_1' , further “up” the long-run land-supply curve where the supply is closer to the limit Q_{MAX} and hence is relatively inelastic. Then, when the biofuels program ends and demand contracts by $-\Delta D$, the demand curve will shift to D_0' and the quantity of land cultivated will change by $-\Delta Q'$. However, even though (by definition) the demand contraction $-\Delta D$ has the same absolute value as the initial demand expansion ΔD , the decrease in cultivated land due to the end of the program ($-\Delta Q'$) generally will be less (in absolute value) than the increase in cultivated land (ΔQ) at the start of the program, because the decrease occurs in a region of less elastic land supply.

Thus, general theoretical considerations suggest that the amount of land reversion at the end of the biofuels program ($-\Delta Q'$ in Figure 4) may be less than the initial land expansion at the beginning of the program (ΔQ in Figure 4). Put less formally, the land-use changes wrought by the initial expansion of demand will not necessarily be *exactly* reversed by the end of the program. For the purpose of the discussion here, however, one wants to know if there is any empirical or theoretical basis for assuming that generally the reversion will be so small relative to the expansion that it reasonably can be ignored.

Theoretically, if rising demand for agricultural commodities and diminishing availability of land over the period of the biofuels program consume nearly all available arable land worldwide such that when the biofuels program ends, if increases in agricultural output are due entirely to increasing inputs on already cultivated land (van Meijl et al., 2006), and if there are no policies encouraging restoration of recently cultivated native ecosystems, then the slackening of demand due to the end of the biofuels program will reduce the input intensity but not the land extent of cultivation. In terms of Figure 4, if D_1' and D_0' cross the long-run supply curve S where it is near Q_{MAX} and hence almost vertical, then shifts in demand will cause large changes in price (and input intensity) but only very small changes in land area ($\Delta Q'$ will be close to zero).⁷

However, these conditions – there is no more land for cultivation, all increases in crop output are due to increases in inputs, and there is no interest in restoration of original ecosystems – are not likely to obtain to a degree sufficient to warrant ignoring

⁷ The opposite situation is that if no relevant supply or demand functions change between the beginning and the end of a biofuels program, then, in terms of Figure 4, D_1' (the demand curve just prior to the end of the program) will be the same as D_1 (the demand curve just after the start of the program), the contraction of demand due to the end of the biofuels program will move the demand curve back to D_0 , and $\Delta Q'$ (the change in land use due to the contraction of demand) will equal $-\Delta Q$. More generally, $\Delta Q'$ will equal $-\Delta Q$ if the supply curve is linear between D_0 and D_1' .

the land-use reversion issue altogether. I discuss each of these next. Another factor that will tend to slightly suppress the amount of reversion in the short-run – the elasticity of the short-run supply curve relative to the elasticity of the long-run supply curve of Figure 4 – is discussed in Appendix B.

First, a simple calculation comparing the rate of expansion of cultivation with the remaining land area suitable for cultivation suggests that the world will not run out of arable land for several hundred years. Statistics from the Food and Agricultural Organization (FAO) of the United Nations indicate that in the year 2000, the total amount of land in cultivation was 11.7% of the total land area of the world, and the total amount of agricultural land, grassland, shrubland, woodland, and forest land *not* already being cultivated was about 68% of the total land area in the world (www.fao.org/forestry/site/fra/en/, www.fao.org/statistics/yearbook/vol_1_1/). However, this 68% includes a substantial amount of boreal forests, dry lands, steep lands, and other areas that are fundamentally unsuitable for cultivation. In this regard, Fischer et al. (2006) perform a comprehensive analysis of the suitability of land worldwide for cultivation, and estimate that about 25% of the global land area not already in cultivation is rain-fed and at least moderately well suited for cultivation. Allowing that irrigation would make additional land suitable for cultivation, I assume that 30% of the uncultivated global land area is potentially cultivatable. Assuming further that the area under cultivation will increase at a rate of 0.2% per year (the rate from 1980 to 2000, according to FAO statistics [www.fao.org/statistics/yearbook/vol_1_1/], and the rate projected by Bouwman et al. [2005] for the period 1995 to 2030), it will take about 600 years to approach the assumed global limit of arable land. (For definitions of land uses in the FAO statistics, see www.fao.org/es/ess/os/envi_indi/annex2.asp.)

Second, intensification of production on existing lands usually results from increased *productivity* of inputs, due to improvements in technology and operation, as well as from increased *use* of inputs, and technological efficiency improvements, unlike greater use of inputs, are not reversed. (However, while technological advancements per se are not reversed by lower prices, the *rate* of technological change *is* affected by prices, such that when demand contracts and prices fall at the end of the biofuels program, the rate of technological change will decline.) That is, to the extent that the suppliers respond to increased demand by developing more productive technologies and processes that increase output without a concomitant increase in inputs (fertilizer, water, seeds, pesticides, etc.), then when demand slackens due to the end of the biofuels program, farmers will respond not by reverting to less productive technologies and less efficient processes, but rather by cultivating less land.

Third, as undisturbed ecosystems become more scarce they become more precious, and society presumably becomes more eager to encourage restoration of original ecosystems wherever possible.

The upshot, then, is that there is no conceptual or theoretical/empirical basis for entirely ignoring the reversion of land uses and associated climate impacts at the end of the biofuels program. Nor is there a strong practical basis, because the models and methods used to estimate the land-use changes associated with the demand shift at the start of the program can be used to estimate the land use changes associated with the demand contraction at the end of the program. However, modeling land-use changes at the end of the program *does* require an additional step: modeling changes in production, consumption, and land use between the start of the biofuels program and the end, in order to establish the supply and demand conditions just prior to the end of program.

In terms of Figure 4, this means modeling the shift in demand from D_1 to D_1' and characterizing the land-supply curve S in the region of D_1' .

Summary of recommended treatment of future versus present impacts and end-of-cultivation issues

For the reasons given above, I believe that it is most reasonable to count the sequestration impacts at the end of cultivation and to apply a continuous discounting function to future impacts. In the LEM, this is done as part of a net-present-value (NPV) calculation (approach #1 in Table 4), in which the model estimates the NPV of the impacts of the actual streams of C emissions due to LUC and then annualizes the NPV (i.e., converts the NPV to an annuity) over the assumed life of the crop-to-energy program. With this NPV method, CO₂ emissions impacts from the initial LUC disturbance are at least partially offset by the C sequestration impacts that occur at the at the end of the bioenergy program when the land-uses revert to their original conditions. (Note that in this framework, “equivalency” is based on the present value of the impacts of climate change.) As a result, the LEM estimates lower CO₂ equivalent emissions from LUC than have studies that implicitly or explicitly use approach #4 of Table 5 (DeLuchi, 1993b; Concauwe et al., 2006; Righelatto and Spracken, 2007; Searchinger et al., 2008; Fargione et al., 2008).

Conclusion

Although much has been written over the past 20 years on the estimation of CO₂ emissions from soils and biomass due to LUC in the lifecycle of biofuels, and several detailed analyses have been performed very recently, some basic conceptual issues have not been addressed adequately. Two issues are particularly important: when, where, and to what extent will land use revert to its original conditions when the bioenergy program ends and cultivation is abandoned, and how in general should society value future impacts relative to present impacts? Estimates of CO₂-equivalent emissions from the lifecycle of bioenergy are quite sensitive to the answers to these questions. With the net-present-value (NPV) method discussed here, in which one counts the impacts of land-use reversion at the end of the bioenergy program and applies a continuous discounting function to future impacts, the impacts of CO₂ emissions from the initial LUC are at least partially offset by the impacts of CO₂ sequestration from reversion. However, this NPV approach is complicated, and introduces additional issues concerning the treatment of the discount rate (e.g., should the discount rate be constant, or should it change over time), emission profiles over time (e.g., do CO₂ emissions from soil follow an exponential decay pattern, as assumed here?), the lag between changes in concentration and changes in temperature, the duration of the biofuels program, the implementation of the approach in biofuel/climate-change policies, and more. Further research on these conceptual, methodological, empirical, and practical issues is needed.

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APPENDIX A. TIPPING POINTS, IRREVERSIBLE IMPACTS, CATASTROPHIES, AND COST-BENEFIT ANALYSIS AND THE DISCOUNT RATE.

As mentioned in the text, some of my colleagues have suggested that the possibility of “tipping points” in climate change – relatively rapid, large scale, irreversible effects such as the melting of large ice sheets or abrupt changes in ocean circulation patterns (see Lenton et al., 2008; Hansen et al., 2007b, 2008; Stern, 2006; and Wright and Erickson, 2003) – confounds the use of a discount rate as outlined here. I believe that closer analysis of this issues shows that this is not so.

To begin, let us distinguish the “ordinary” impacts of climate change from the impacts associated with the afore-mentioned tipping points. The tipping-point impacts ostensibly are much more uncertain, much larger (in physical terms), more rapid, and less easily reversed than are the “ordinary” impacts. (In Lenton et al. [2008], a key feature of a tipping point is that a small change in some parameter in the climate system can lead to a qualitative change in some crucial aspect of the climate system.) Let us also distinguish the *theoretical* from the *practical* applicability of cost-benefit analysis. Now, given that there have been many damage-cost analyses of the ordinary impacts of climate change (for summaries and reviews, see Kuik et al., 2008; IPCC, 2007; Stern, 2006; AEA Technology Environment, 2006; Delucchi, 2006; and Tol, 2005), and that there is no *theoretical* reason that tipping-point damages cannot be estimated (because in theory cost-benefit analysis can include everything), the question before us is whether the impacts associated with “tipping points” are so qualitatively different from the “ordinary” impacts of climate change that even if one can estimate the ordinary impacts, one nevertheless cannot *credibly* estimate tipping-point impacts.

For several reasons, I believe that tipping-point impacts are *not* so much different from ordinary impacts as to make it practically impossible to estimate them credibly. First, unless the “tipping points” are like the asteroid strike at the K-T boundary—a complete surprise and an instant global disaster—humans will have plenty of time to prepare, and hence will be able to adapt to or endure the “catastrophe” over many decades, rather than over hours and days. Under no circumstances will sea level 5 meters overnight, regardless of what one thinks will happen to the Greenland and Anarctic ice masses. It will rise over many decades, and we will know about it well in advance (Hansen et al., 2007a). This sort of scenario, while global and large, can *in principle* be described and valued with the tools of ordinary cost-benefit scenario analysis. The Tol et al. (2006) analysis of adaptation to five metres of seal level rise and Keller et al. (2004) analysis of the economic implications of the collapse of the ocean thermohaline circulation system point the way to this sort of cost-benefit scenario analysis. (See also Stockholm Environment Institute [2005] and Wright and Erickson [2003].)

Second, society cares ultimately about the magnitude of the *costs* of climate change, and it is not necessarily the case that the “largest” physical impacts generate the largest costs. In cost-benefit analyses of climate change, some of the largest costs arise from people dying from heat or disease (Bosello et al., 2006; see Patz et al. [2005] for a discussion of the impacts of climate change on human health), and it is not immediately obvious that, for example, large changes in sea level over many decades will significantly increase the number of people dying from heat or disease. Bosello et al. (2006) and Tol’s FUND model (Warren et al., 2006) estimate the health costs of climate change directly as a function of temperature. Moreover, it is reasonable to expect that society will prefer to sharply reduce its vulnerability to extreme climate change events

rather than suffer the unmitigated impacts (e.g., Pielke, 2007a, 2007b), and presumably mitigation costs much less than does the unmitigated damage.

Third, there is nothing particularly problematic about estimating the costs of irreversible impacts, for this is done routinely in cost-benefit analysis generally and in cost-benefit analysis of climate change specifically (e.g., Warren et al., 2006; Keller et al., 2004). “Irreversible” does not mean “impossible to value” or “of infinite value,” and most basic treatments of cost-benefit analysis present non-market valuation techniques for estimating the value of “irreversible” impacts, such as species extinction. (For a general discussion of issues in ecosystem valuation, see Nijkamp et al. (2008) and the special issue “The Dynamics and Value of Ecosystem Services: Integrating Economic and Ecological Perspectives,” *Ecological Economics* 41 (2) (2002).

Fourth, although the timing and impacts of tipping points are very uncertain – even more uncertain than “ordinary” impacts – this presents a fundamentally intractable analytical problem only if the uncertainty is infinite (Tol, 2003). Tol (2003) suggests that even in climate-change scenarios with essentially “infinite” uncertainty, the *difference* between the scenario and the baseline – which is the relevant quantity – may well be finite. Yoh (2003) and Howarth (2003) comment on Tol’s (2003) analysis and conclude that it does not demonstrate the potential inapplicability of cost-benefit analysis. Furthermore, in a later paper that discusses uncertainty in estimates of marginal damage costs, Tol (2005) says:

Actively working in the area of external costs of energy in general and climate change in particular, I am often confronted with people who argue that climate change is too uncertain to say anything about the marginal damage costs of carbon dioxide emissions. The uncertainties are indeed substantial, but not as large as these people think (p. 2072).

And most recently, Tol and Yohe (2007) address the issues raised in Tol (2003) and show that “a portfolio of international policies with at least two independent tools can avoid infinite uncertainty at the margins...even in the relatively unlikely event that climate change causes negative economic growth in a region or two” (p. 429).

Finally, it is quite possible that a marginal change in GHG emissions merely will *shift* the timing of tipping points, perhaps by a few decades at most. Given that the climate-change damage cost of any emission scenario is the *difference* between the damage-cost profile with the emission scenario and the damage-cost profile without the scenario, then if the emission scenario merely shifts the tipping point, the cost attributable to the scenario is not the total amount of damage past the tipping point, but rather the difference between the present value of the damages after the “shifted” tipping point and the present value of the damages after the “unshifted” tipping point.

APPENDIX B. THE AMOUNT OF LAND REVERSION IN THE SHORT RUN.

Figure 4 and the associated discussion of reversion are based on a long-run land-supply curve. In the short run there can be an additional factor that can slightly reduce the amount of land reversion compared with the long run. The short-run supply curve is less elastic than the long-run supply curve, and as a result in the short run a change in demand can cause a larger price change and a smaller quantity change than in the long run. However, if it occurs at all, this short-run supply-curve effect is likely to be small and brief.

The short-run supply curve differs from the long-run supply curve because in the short run some costs are “fixed” or “sunk” and hence can’t be saved. The greater the share of fixed costs of the total, the greater the difference between the short-run and the long-run supply elasticity. However, in the U. S., fixed costs appear to be a small fraction of total farming costs: according to National Agricultural Statistics Service forecasts of the components of the total cost of production for various crops (www.ers.usda.gov/Data/CostsAndReturns/data/Forecast/cop_forecast.xls), “fixed costs” are probably 15% to 30% of total costs of production. (I counted as a “fixed cost” all capital recovery costs, 15% of land costs, and 20% of general farm overhead costs.) This means that the difference between the short-run supply elasticity and the long-run supply elasticity is relatively small.

Any short-run effects also will be brief. We are in the short run only for a little while; in the long-run, all costs are variable. Eventually, capital needs to be replaced or repaired. It is even possible that if a price change, due to a demand contraction, is recognized as permanent, capital will be sold or put to other uses immediately.

Finally, there will be a short-run effect only if the contraction of demand affects production on agricultural land that is already developed and hence has some fixed costs that cannot be saved in the short run by idling agricultural production. Put another way, there will be a short-run effect only if the land affected is characterized by the short-run supply curve. But it is likely that the land affected will not be developed already, and hence will be characterized instead by the long-run supply curve. Demand for agricultural commodities generally expands over time. To the extent that this ongoing expansion is met by bringing new, undeveloped land into production, a one-time contraction of demand (as in Figure 4) will have the effect of foregoing agricultural expansion into new, undeveloped land. Since by definition the development of new, previously undeveloped land requires the full investment up front, to forego development of this new land is tantamount to reversion along the full-cost – or long-run – supply curve.

TABLE 1. SIX COMPONENTS OF AN IDEAL MODEL OF C EMISSIONS DUE TO LAND-USE CHANGE

Component in an ideal model	Treatment in the LEM
1. A relationship between changes in bioenergy production and changes in land use.	Simple parameterization.
2. A relationship between changes in land use and changes in CO ₂ emissions from soils and plants.	Representation of soil and plant CO ₂ emissions over time.
3. A relationship between changes in C emissions and changes in atmospheric CO ₂ levels.	Multiple CO ₂ “decay” streams.
4. A relationship between changes in atmospheric CO ₂ and changes in climate.	Representation of relationship between changes in CO ₂ and changes in radiative forcing and temperature.
5. A relationship between changes in climate and changes in impacts on people and ecosystems.	$\Delta D = f(\Delta T^K)$, where ΔD is the change in damages, ΔT is the change in temperature, and K is an exponent (usually between 1.5 and 2).
6. A method for valuing and aggregating climate impacts over time, and for making the time stream of climate impacts due to LUCs commensurate with the time stream of climate impacts due to other parts of the lifecycle of bioenergy.	Comprehensive present-value/annualization method.

LEM = Lifecycle Emissions Model (unpublished update of Delucchi [2003]).

TABLE 2. HIERARCHY OF STREAMS IN THE REPRESENTATION OF THE CLIMATE IMPACTS OF BIOENERGY POLICIES

Stream in the real world	Treatment in the LEM
1. <u>Program actions</u> . Prices, yields, supply curves, and land uses can change over time, year-by-year, in the “with bioenergy program” case compared to the “no bioenergy program” case.	LEM represents changes in a single “target year” (TY) only. (The LEM can be run for different TYs, but it does not aggregate results over TYs.)
2. <u>Emissions</u> . Then, each change in land use (in each year) generates its own time series of changes in carbon emissions; for example, a change in land use in any year T initiates a process of carbon emission or sequestration that can continue for many years after T .	Explicit representation of duration and shape of soil-C and plant-C emission streams (Figure 2).
3. <u>Concentration and radiative forcing</u> . Next, each change in carbon emission or sequestration (in each year) generates its own time series of changes in CO ₂ concentration (atmospheric C stocks) and radiative forcing; for example, an emission of carbon from soils in year $T+x$ (due ultimately to LUC in year T) will generate an atmospheric CO ₂ concentration and decay profile and associated radiative-forcing effects that extend for many decades beyond $T+x$.	CO ₂ decay and radiative forcing represented explicitly as part of estimation of CO ₂ -equivalency factors; see Figure 3a here for an illustration of the relationship between emissions changes (stream #2) and concentration changes (stream #3).
4. <u>Climate (temperature) change</u> . Next, any change in radiative forcing in any year will generate a stream of climate changes, the lag between radiative forcing (stream #3) and climate change (stream #4) being due mainly to the thermal inertia of the oceans.	Thermal inertia lag between radiative forcing and climate change represented explicitly; see Figure 3a for illustration.
5. <u>Impacts</u> . Finally, any change in climate, in any year (stream #4), can impact people and ecosystems for many years (e.g., by changing the incidence of chronic diseases).	The LEM does not formally represent impact streams.

LEM = Lifecycle Emissions Model (unpublished update of Delucchi [2003]).

TABLE 3. PARAMETERS IN THE SIMPLE MODEL (EQ. 1) USED TO GENERATE FIGURE 3A.

Flux or stock	Initial value (C_0)	e-fold time (L)	Comment
Soil-C net loss	0.25 kg-C/m ² /yr. in year 1	20 years	Results in total loss of 4.6 kg-C/m ² over 50 years.
Native plant C loss	0.5 kg-C/m ² /yr. in year 1	2 years	Results in total loss of 1 kg-C/m ² .
Bioenergy-crop system C buildup	-0.25 kg-C/m ² /yr. in years 1,2	see comment	Assume constant rate of C build up in whole system (including fuel reserve) in first two years.
Soil-C net recovery	-0.19 kg-C/m ² /yr. in year 50	25 years	Initial value is amount that, given e-fold time, results in amount of soil C recovery equal to the total soil C loss over the cultivation period (50 years).
Native plant C regrowth	-0.10 kg-C/m ² /yr. in year 50	10 years	Initial value is amount that, given e-fold time, re-sequesters the total C originally lost.
Bioenergy-crop system C loss	0.25 kg-C/m ² /yr. in years 50, 51	see comment	Assume constant rate of system C loss over two years.
Land C stock	normalized to 0 at start of year 1	n.a. – see comment	Stock in year T is stock in year $T-1$ minus soil-C, plant-C, and bioenergy crop-system-C losses, plus soil-C recovery, plant-C regrowth, and bioenergy-crop-system C buildup.
Atmosphere C stock	normalized to 0 at start of year 1	82.4% of CO ₂ has 93.5-yr. life; 17.6% has infinite life	Stock in year T is stock in year $T-1$ plus the amount of C remaining in year T from <i>each</i> decaying net emission stream from year 0 to year T , where the net emission in any year is the change in the land-C stock.
Ocean C stock	normalized to 0 at start of year 1	n.a. – see comment	Stock in year T is the difference between the total land+air+ocean C stock at start of year 1 and the land and air C stocks in year T .
Atmospheric temperature change	0 at start of year 1	50 years for lag time	Temperature change in year T is 98% of change in year $T-1$ plus 2% of the ultimate potential change due to forcing in year T . (Assume here that radiative forcing per unit C, and climate sensitivity to forcing, are effectively constant.)

Note that these values are meant to be broadly representative of the effects of cultivating temperate grasslands (e.g., Houghton and Hackler, 2001).

TABLE 4. FOUR APPROACHES TO ESTIMATING CO₂-EMISSION IMPACTS DUE TO LAND-USE CHANGE, ACCORDING TO HOW ONE TREATS EOC EMISSIONS AND HOW ONE VALUES FUTURE IMPACTS RELATIVE TO PRESENT IMPACTS

	<i>Count sequestration after cultivation ends</i>	<i>Do not count sequestration after cultivation ends</i>
<i>Discount all future impacts</i>	1. Estimate the NPV of emission impact streams, including EOC impacts, then annualize the NPV over the life of the bioenergy program. This is the approach recommended here.	2. Same as approach #1, except ignore EOC sequestration impacts on the grounds either that cultivation won't end or else that ending cultivation is a separate policy decision. As discussed in the text, these grounds for ignoring EOC impacts are not defensible.
<i>Do not discount anything up to some time horizon TH</i>	3. Estimate initial emission impacts plus EOC impacts divided by fuel output over life of the bioenergy program (LP_E), where $LP_E \leq TH$ and the EOC impacts are negative emissions (sequestration). I am not aware of anyone advocating this approach.	4. Estimate initial emission impacts divided by fuel output over the lesser of LP_E or TH, where we ignore EOC sequestration impacts either because of the (indefensible) reasons given in approach #2 or because $LP_E > TH$. This approach is implicit in estimates of the "payback period" for initial LUC emissions (e.g., DeLuchi, 1993b; Concawe et al., 2006; Righelato and Spracklen, 2007; Reijnders and Huijbregts, 2008; Searchinger et al., 2008; Fargione et al., 2008; Kim et al., 2009; Piñeiro et al., 2009) ^a

NPV = net present value; EOC = end of cultivation; TH = time horizon.

^a Piñeiro et al. (2009) perform a standard "payback" analysis but also analyze the net present value of GHG fluxes under different land-use conversions, as described in the text.

TABLE 5. QUALITATIVE CONTRIBUTION OF CO₂-EMISSIONS FROM LAND-USE CHANGE TO TOTAL CO₂-EQUIVALENT EMISSIONS IN THE LIFECYCLE OF BIOFUELS, ACCORDING TO HOW ONE TREATS EOC SEQUESTRATION AND HOW ONE VALUES FUTURE IMPACTS RELATIVE TO PRESENT IMPACTS

	<i>Count sequestration after cultivation ends</i>	<i>Do not count sequestration after cultivation ends</i>
<i>Discount all future impacts</i>	<p>1. <u>Contribution:</u> ~ 0 to ++</p> <p><u>Comment:</u> Depends on assumptions about the amount and type of native land disturbed by cultivation, the discount rate profile, the duration of the bioenergy program, the characteristics of C emission streams, and more.</p>	<p>2. <u>Contribution:</u> ~ 0 to ++++</p> <p><u>Comment:</u> This generates higher net emissions than does #4 because the amortized initial emission due to LUC at program start increases with the discount rate.</p>
<i>Do not discount anything up to some time horizon TH</i>	<p>3. <u>Contribution</u> ~ 0 to +</p> <p><u>Comment:</u> If EOC emissions are close to or equal to initial emissions and of the opposite sign, this method will result in small or zero g-CO₂/KJ-fuel emissions due to LUC.</p>	<p>4. <u>Contribution:</u> ~ 0 to +++</p> <p><u>Comment:</u> Depends on assumptions about the amount and type of native land disturbed by cultivation, and other factors.</p>

EOC = end of cultivation. *Source:* author estimates based on unpublished updates to the Lifecycle Emissions Model (LEM; see Delucchi [2003] for documentation of an earlier version of the LEM).

FIGURE 1. SCHEMATIC REPRESENTATION OF THE EFFECTS OF BIOENERGY POLICIES ON CLIMATE AND CLIMATE IMPACTS

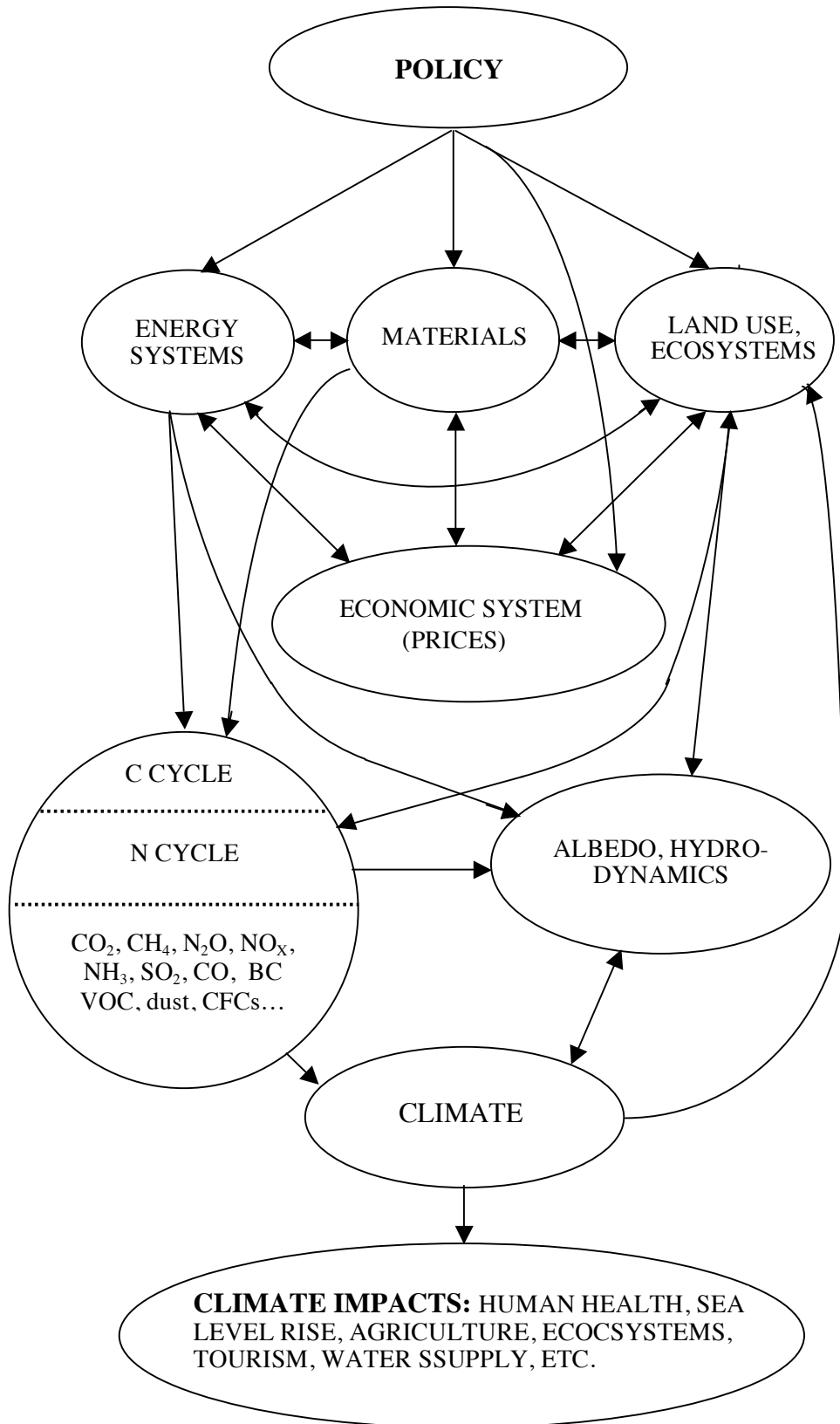


FIGURE 2. C EMISSIONS FROM PLANTS AND SOILS DUE TO LAND-USE CHANGE

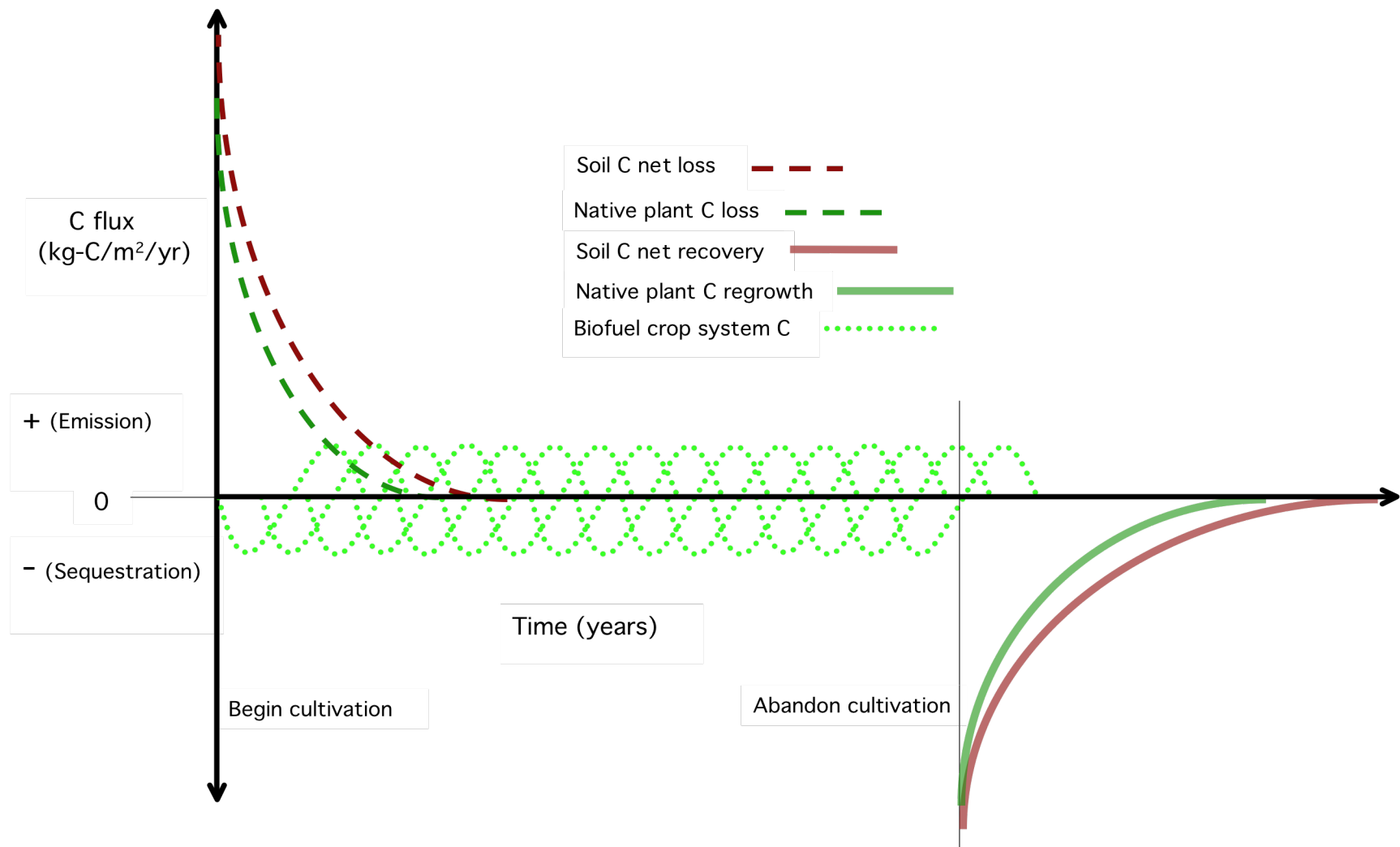


FIGURE 3A. CHANGES IN LAND, ATMOSPHERE, AND OCEAN C STOCKS DUE TO C EMISSIONS FROM LAND-USE CHANGE (WITH REVERSION OF LAND USE AT END OF CULTIVATION)

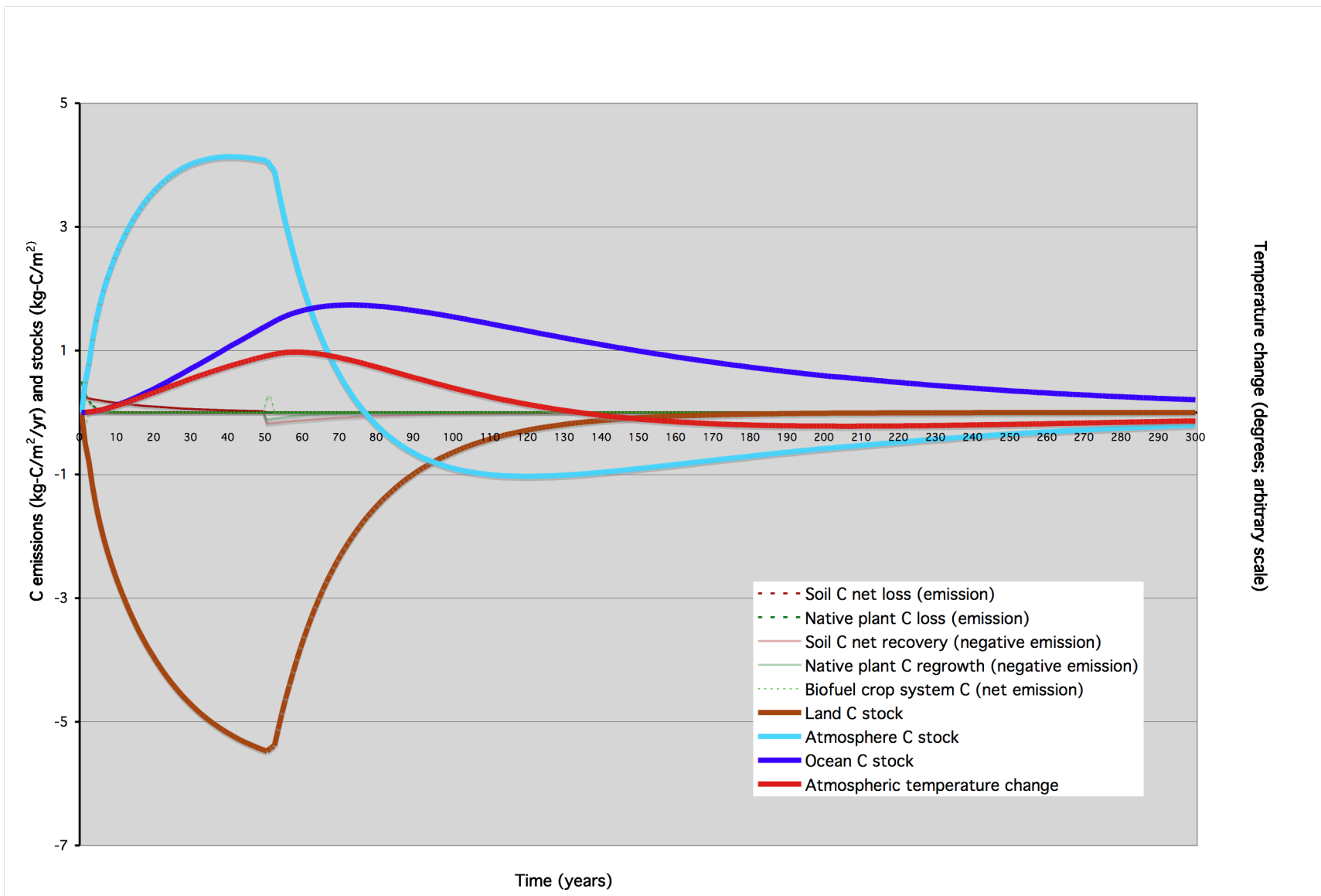


FIGURE 3B. CHANGES IN LAND, ATMOSPHERE, AND OCEAN C STOCKS DUE TO C EMISSIONS FROM LAND-USE CHANGE (NO REVERSION OF LAND USE AT END OF CULTIVATION)

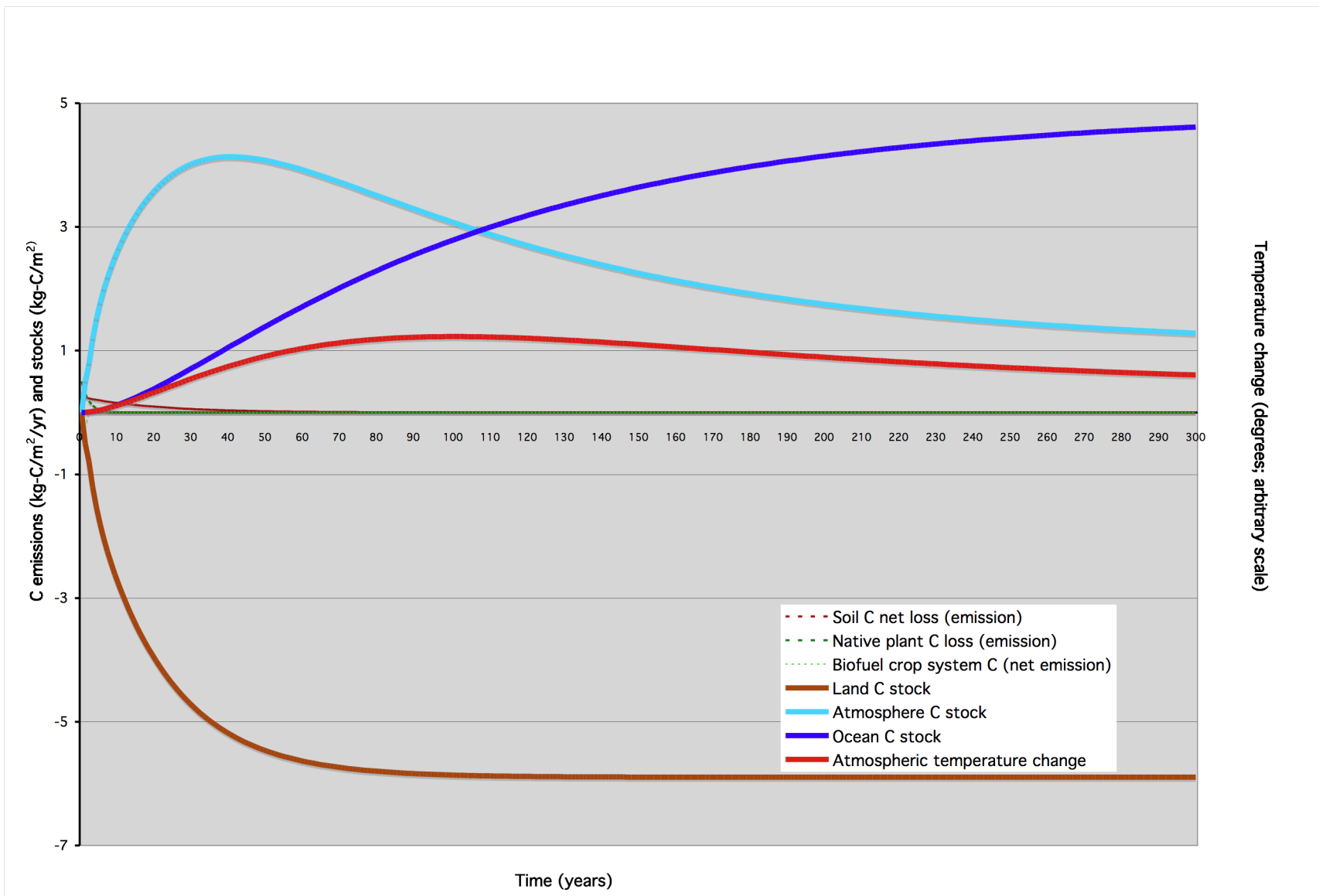
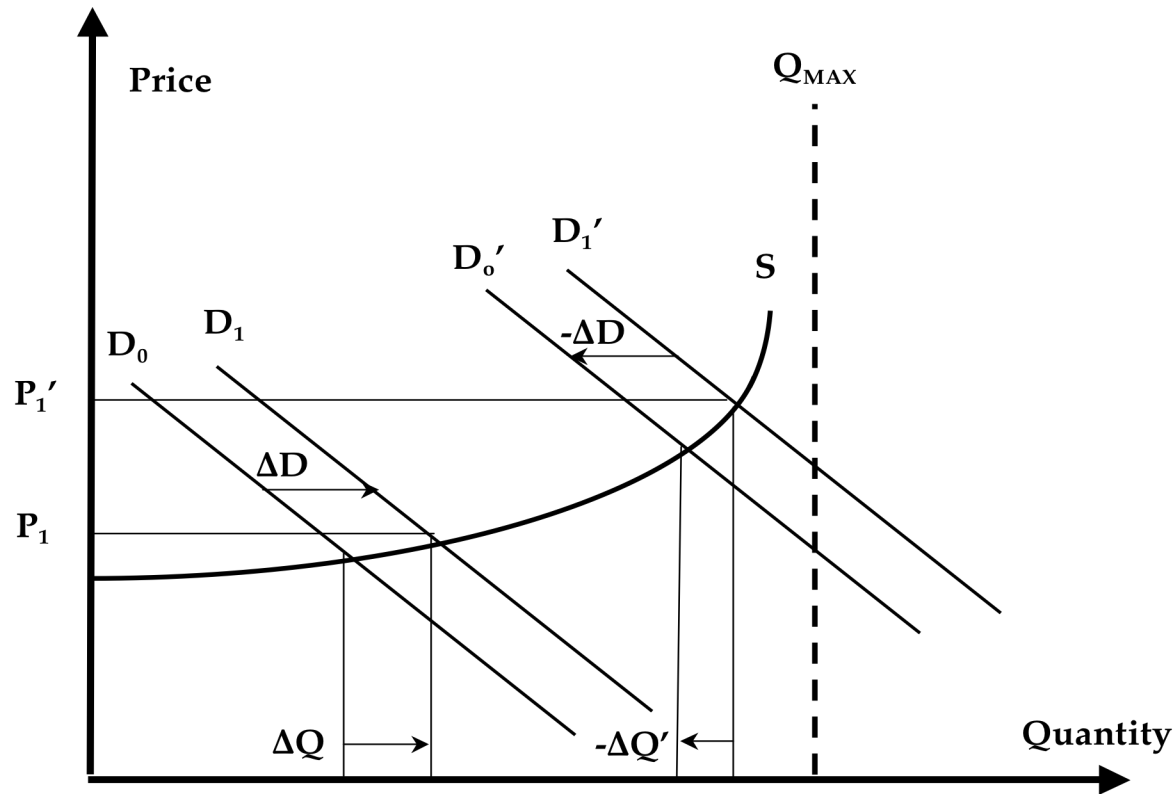


FIGURE 4. LAND-USE CHANGES AT THE START AND THE END OF A BIOFUELS PROGRAM



D_0 = demand curve just prior to start of biofuels program; D_1 = demand curve just after start of biofuels program; D_1' = demand curve just prior to end of biofuels program; D_0' = demand curve just after end of biofuels program; S = supply curve; ΔD = expansion of demand due to start of biofuels program; $-\Delta D$ = contraction of demand due to end of biofuels program; ΔQ = increase in quantity of land cultivated due to expansion of demand for biofuels; $-\Delta Q'$ = decrease in quantity of land cultivated due to contraction of demand for biofuels; Q_{MAX} = maximum amount of land available for cultivation; P_1 = price just after start of biofuels program; P_1' = price just prior to end of biofuels program.