Using the Lashof Accounting Methodology to Assess Carbon Mitigation Projects With Life Cycle Assessment

Ethanol Biofuel as a Case Study

Alexandre Courchesne, Valérie Bécaert, Ralph K. Rosenbaum, Louise Deschênes, and Réjean Samson

Keywords:

carbon sequestration
climate change
environmental assessment
industrial ecology
life cycle assessment (LCA)
radiative forcing

Summary

As governments elaborate strategies to counter climate change, there is a need to compare the different options available on an environmental basis. This study proposes a life cycle assessment framework integrating the Lashof accounting methodology, which enables the assessment and comparison of different carbon mitigation projects (e.g., biofuel use, a sequestering plant, an afforestation project). The Lashof accounting methodology is chosen amid other methods of greenhouse gas (GHG) emission characterization for its relative simplicity and capability to characterize all types of carbon mitigation projects. Using the unit of megagram-year (Mg-year), which accounts for the mass of GHGs in the atmosphere multiplied by the time it stays there, the methodology calculates the cumulative radiative forcing caused by GHG emission within a predetermined time frame. Basically, the developed framework uses the Mg-year as a functional unit and isolates impacts related to the climate mitigation function with system expansion. The proposed framework is demonstrated with a case study of tree ethanol pathways (maize, sugarcane, and willow). The study shows that carbon mitigation assessment through life cycle assessment is possible and that it could be a useful tool for decision makers, as it can compare different projects regardless of their original context. The case study reveals that system expansion, as well as each carbon mitigation project's efficiency at reducing carbon emissions, are critical factors that have a significant impact on the results. Also, the framework proves to be useful for treating land-use change emissions, as they are considered through the functional unit.

Address correspondence to:
Réjean Samson
CIRAIG—Department of Chemical Engineering
École Polytechnique de Montréal
P.O. Box 6079 Downtown
Montreal (Quebec) Canada H3C 3A7
rejean.samson@polymtl.ca
www.ciraig.org

DOI: 10.1111/j.1530-9290.2010.00228.x

Volume 14, Number 2

Background, Aim, and Scope

Many carbon mitigation options are possible to reduce atmospheric greenhouse gas (GHG) concentrations, and the fight against climate change needs diversified actions. According to Lal (2008), three strategies enable atmospheric carbon mitigation: carbon sequestration (geological or biological), utilization of low-carbon fossil fuel, and energy efficiency measures.

These methods either reduce atmospheric carbon or lower fossil carbon emissions. Biofuel, a low-carbon fossil fuel, transforms biogenic carbon from plants into transportation fuel, thus permitting a shorter carbon cycle within the atmospheric and biotic carbon pool. Because they add less fossil carbon to the atmosphere, biofuels are considered to have the potential to mitigate climate change.

Unfortunately, carbon-mitigating projects can generate negative impacts in other environmental spheres (Khoo and Tan 2006b; Zah et al. 2007; Lal 2008). Biofuels, for example, have potential environmental drawbacks that arise from agriculture, transport, transformation, and land-use change (LUC) emissions (Fu et al. 2003; Sheehan et al. 2003; Kim and Dale 2006; Fargione et al. 2008; Searchinger et al. 2008; Piñeiro et al. 2009). LUC emissions are the GHGs released from the vegetation and the soil of land cleared for agricultural use. Hence, it is important to measure those environmental consequences to help decision makers choose the best options and avoid any important impact displacement.

Researchers can take various approaches, such as carbon footprinting, to calculate the carbon-reducing impact of a given project, but a broader approach is needed to characterize a large number of environmental impacts. Life cycle assessment (LCA) is a standardized tool that considers the full life cycle (cradle to grave) to assess the environmental impacts of any specific product or service (ISO 2006a, 2006b). It is already used to analyze mitigation projects. For example, Khoo and Tan (2006a) compare the impact of a carbon sequestration power plant with that of a typical power plant, and Kim and Dale (2006) compare biofuel with petroleum as used to fuel a car. To date, however, life cycle studies consider carbon reduction as a benefit and not as a function. This confines the study to the "typical" function of the product (to produce electricity or power a car) and impedes broader comparison. How may one decide which carbon mitigation project is most environmentally interesting?

LCA uses the functional unit to establish the basis on which each studied option will be compared. The choice of the functional unit is critical, as it must have the capacity to accommodate all possible projects, must be of accessible simplicity for its wide diffusion, and must also be pertinent to decision makers. For a carbon mitigation project, the function should be in the form of climate change impact reduction, such as a slower temperature increase, or in the form of a climate change cause reduction, such as a lower radiative forcing (RF) from GHGs. Although attempts have been made (Kirschbaum 2003), evaluating the beneficial impacts of carbon-mitigating projects is complex, because many interactions are involved. The science linking carbon mitigation actions and true climate change relief is still subject to a great deal of uncertainty (IPCC 2007b).

Another option is to compare different mitigation projects on the basis of a reduction in radiative forcing, which is the main cause of climate change (IPCC 2007b). There exists no consensus in the literature as to how to allocate carbon credit to the different types of carbon mitigation projects. Existing calculation methods include global warming potential (GWP); megagram-year (Mg-year) accounting, which addresses either atmospheric carbon (Lashof method) or sequestered carbon (Moura-Costa method; Fearnside 2000; Fearnside et al. 2000; Moura Costa and Wilson 2000). Yet another option is the use of atmospheric models, such as Bern (Joos et al. 2001) or REFUGE (Korhonen et al. 2002), that directly calculate atmospheric carbon evolution.

GWP is a measure of the relative radiative forcing caused by any non—carbon dioxide (non-CO_2) gas over its lifetime compared to the radiative forcing caused by an equivalent amount of CO_2 (IPCC 2007b). GWPs are calculated for 20-year, 100-year, and 500-year time horizons and are expressed in mass of CO_2-equivalents. The underlying model includes basic atmospheric reactions, such as degradation, and interaction with other carbon pools, such as oceans. Although GWPs are good at characterizing GHG emissions, they are problematic for quantifying carbon sequestration, especially temporary sequestration, because GWPs do not differentiate emissions released now from those released in the future (IPCC 2000). This temporal aspect is present in many mitigation projects, as they can have an irregular GHG release over time. Afforestation, for example, sequestrates carbon in the growth phase but also emits carbon during the transport and planting phases. Furthermore, actual mitigation can be erased later if the plantation is cut or burned (IPCC 2000).

Atmospheric models, such as Bern (Joos et al. 2001) or REFUGE (Korhonen et al. 2002), have the advantage that they calculate the GHG concentration, which is directly proportional to radiative forcing. Unfortunately, such models are complex and not available to most researchers.

Mg-year atmospheric carbon-accounting methodologies calculate the radiative forcing that is avoided by a mitigation project on a year-to-year basis. This approach is similar to GWP, as it can lead to the exact same result, but its dynamic calculations allow the comparison of temporary sequestration, permanent sequestration, and fossil carbon emission reduction. In this study, we chose the Lashof methodology for the determination of the functional unit. Although it does not directly analyze atmospheric carbon evolution, it has the advantage of including a dynamic approach and a carbon-decaying function of relative simplicity to model atmospheric carbon decrease. Also, it allows one to determine a date by which results are calculated. This is important for decision makers, as they can evaluate the benefits from a project in the time frame of their choice.

The main objective of this study is to propose a tool that allows researchers to compare the environmental impacts of different carbon mitigation projects that produce the same amount of carbon mitigation in a predetermined time frame. To assess the environmental impacts of the project, we chose LCA, because it includes a broad range of environmental indicators and is an internationally recognized methodology. The framework uses a cumulative radiative forcing (CRF) reduction indicator for the functional unit (1 megagram carbon dioxide-equivalent year [Mg CO_2eq-year]),[1] calculated with the Lashof accounting methodology. We chose this unit because it has a broader applicability to characterizing carbon mitigation projects than does customary GWP. To test the framework, we investigate a case study of three ethanol scenarios and compare the results from a typical transport LCA with the results of a carbon mitigation LCA. We also explore the effect of including potential LUC emissions.

Methodology

Megagram-year (Mg-year) Accounting

Radiative Forcing and GHG Decay

The Lashof method evaluates the CRF of each GHG over a period of time. When a GHG emission occurs, it generates radiative forcing in the 1st year. Afterward, it interacts with nature, and its initial quantity decreases over time. Hence, the impact on climate change diminishes with the decreasing quantity of GHG in the atmosphere. The CO_2 response function used in this study is based on the revised version of the Bern carbon cycle model and uses a background CO_2 concentration value of 378 parts per million. The response function can model emission and removal of carbon from the atmosphere. The radiative forcing of a pulse emission of CO_2 with time t is given by the IPCC (2007b):

$$RF_{CO_2}(t) = Q_{CO_2}\left[a_0 + \sum_{1}^{3} a_i e^{-t/\tau_i}\right] \quad (1)$$

where RF_{CO_2} represents the radiative forcing in Mg CO_2eq caused in year t by a CO_2 emission emitted t years ago and Q_{CO_2} is the amount of CO_2 emitted at year 0 (in megagrams). As CO_2 is the reference molecule, mass is equal to radiative forcing. The term in brackets (decaying function) represents the remaining fraction of the initial emission at time t (year); $a_0 = 0.217$, $a_1 = 0.259$, $a_2 = 0.338$, $a_3 = 0.186$, $\tau_1 = 172.9$ years, $\tau_2 = 18.51$ years, and $\tau_3 = 1.186$ years (IPCC 2007b).

The decaying function represents the reaction of carbon sinks, mainly ocean, to the increase in atmospheric carbon (Joos et al. 2001). Because carbon emissions increase atmospheric carbon content, the gradient of carbon concentration between the atmosphere and the sinks also increases, raising the transfer from the atmosphere to the sinks. Therefore, an increase of carbon will, over time, be tampered by the carbon sinks. Reversed reactions occur with atmospheric carbon removal (Korhonen et al. 2002).

For other GHGs, the model uses a first-order kinetic decreasing equation for decay and instant radiative properties compared to CO_2. The characteristics used in equation (2) are the lifetime (half-life) and the instant radiative efficiency, which can be found in the latest IPCC report.

$$RF_x(t) = Q_x R_x e^{-t/\tau_x} \tag{2}$$

where RF_x represents the radiative forcing in Mg CO_2eq caused in year t by the emission of gas x emitted t years ago, Q_x is the amount of gas x emitted at year 0, R_x is the instant radiative efficiency of gas x compared to CO_2, and τ_x is the lifetime (half-life) of gas x. For example, methane (CH_4) and dinitrogen oxide (N_2O) have lifetimes of 12 and 114 years, respectively, and instant radiative efficiencies of 72 and 216 compared to CO_2.

The radiative efficiency value of carbon monoxide (CO) remains uncertain. Most of its effect comes from indirect reaction with ozone and hydroxyl (OH) radicals and subsequent oxidation to CO_2 (IPCC 2007b). The IPCC gives CO a 100-year GWP of 1.9 but also states important regional variations. In this study, which concerns radiative forcing, we treat CO as a CO_2 emission.

The preceding equations are the same as those the IPCC used to calculate GWP factors of non-CO_2 GHG. They can be determined by the ratio of the integrals of equation (2) and equation (1) with a time span of 20, 100, or 500 years.

Mg-year Score

The score calculated by the Lashof method is expressed in Mg-year, which is the sum of each year's radiative forcing over the period of time, written as Mg of CO_2-equivalent radiative forcing multiplied by time. Previous equations calculate radiative forcing of a pulse emission of GHG for any year after the emission. The following equation calculates the radiative forcing of one specific year, which we calculate by adding the radiative forcing of every GHG emitted before the chosen year:

$$RF_k = \sum_{i=1}^{k} RF_{x1}^{i} + RF_{x2}^{i} \cdots \tag{3}$$

where RF_k is the radiative forcing of year k and RF_{x1}^{i} and RF_{x2}^{i} are the radiative forcing caused in year k by gas x_1 and x_2 emitted in year i, respectively. All amounts are expressed in Mg CO_2eq. For example, a 5-year project may be considered for which 1 Mg of CO_2 is emitted each year, and RF_3 needs to be calculated. Equation (3) is then a summation of three terms ($RF_{CO_2}^{1}$, $RF_{CO_2}^{2}$, $RF_{CO_2}^{3}$) representing the radiative forcing caused in year 3 by the emissions that occurred in the 1st, 2nd, and 3rd years of the project, respectively.

We then calculate the Mg-year score by summing each year's radiative forcing:

$$CRF = \sum_{k=1}^{n} RF_k \tag{4}$$

where CRF is the cumulative radiative forcing in Mg-year, n is the length of the predetermined time frame in years and represents the last year for which the radiative forcing is included in the calculation of the CRF in years, and RF_k is the radiative forcing of year k in Mg CO_2eq.

Figure 1 shows the progression of the Mg-year score for an emission of 1 tonne of CO_2, decaying with the Bern model equation. This tonne of CO_2 has a score of 48 Mg-year 100 years after being emitted. The slope of the curve will never reach zero, but slope reduction will discontinue around 500 years due to the properties of equation (1). On the opposite end, if 1 tonne of CO_2 is removed from the atmosphere, the progression of the Mg-year score will yield a curve with a slope that is symmetrical to the first about the x-axis of the graph.

Framework

To use the Mg-year as the functional unit, one must choose a time frame for the reduction. The functional unit is expressed in q Mg-year reduction achieved in n years. The choice of the time frame can have very important impacts on results and must be made carefully. As shown in figure 1, radiative forcing caused by CO_2 steadily

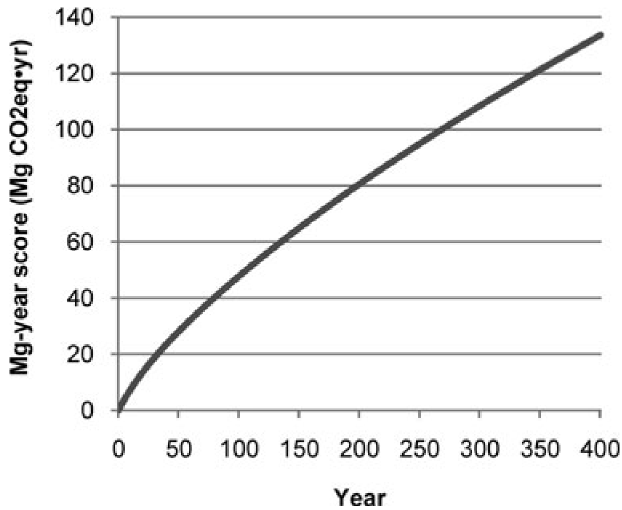

Figure 1 Megagram-year (Mg-year) progression of the emission of 1 tonne of CO_2 at year 0.

increases with time; therefore, one needs a defined time frame to calculate a CRF. This explains the 20-year, 100-year, and 500-year time frames chosen by the IPCC to calculate GWP. In this particular situation, the Mg-year unit is used to characterize carbon mitigation projects. As climate change is happening, a carbon mitigation project worth considering should at least have beneficial impacts within the time horizon of the appearance of its effects. Thus, the time frame could be used as a primary criterion to rule out any carbon mitigation project that does not reduce radiative forcing within a reasonable amount of time. This approach works relatively well with projects that have constant carbon emission reduction (e.g., energy efficiency) but could disadvantage projects that cause increasing carbon reduction. For example, a forest plantation project with annual plantation could attain its maximum sequestering potential years after the beginning of the project, but, because of the GHG-generating plantation activities, it could be ruled out if the chosen time frame is too short to include the main sequestering phase. For further discussion on choosing the time frame to evaluate carbon-mitigating projects, see work by Fearnside (2002).

Some projects, such as afforestation, directly remove carbon from the atmosphere; thus, the Mg-year reduction can be calculated directly from the project data. In such projects, calculations of the CRF lead to negative results; however, as most climate mitigation projects do not remove carbon from the atmosphere, the calculation of a Mg-year reduction needs a comparison with a baseline project. Hence, we first carry out a comparative LCA for the business-as-usual scenario, in which no carbon mitigation project is implemented (petroleum transport), and for the carbon-mitigating project (biofuel) scenario. This LCA is based on the functionality the scenarios share and the function for which the carbon mitigation project replaces the baseline (kilometers, for the biofuel study).[2] Throughout the text, we refer to these LCAs as the "typical" LCA.

The proposed framework (see figure 2) basically adds three steps to the traditional LCA methodology:

1. After the life cycle inventory (LCI) step of the "typical" LCA, GHG data are retrieved from the inventory and compiled with the Mg-year method for the time frame studied. This step yields the Mg-year (or CRF) score for each scenario of the "typical" LCA.
2. The Mg-year scores of the carbon mitigation project and baseline scenarios are compared, and a coefficient (Rc) is calculated according to equation (5):

$$Rc = q_n/(CRF_b - CRF_{cm}) \quad (5)$$

where Rc is the reduction coefficient, CRF_b and CRF_{cm} are the cumulative radiative

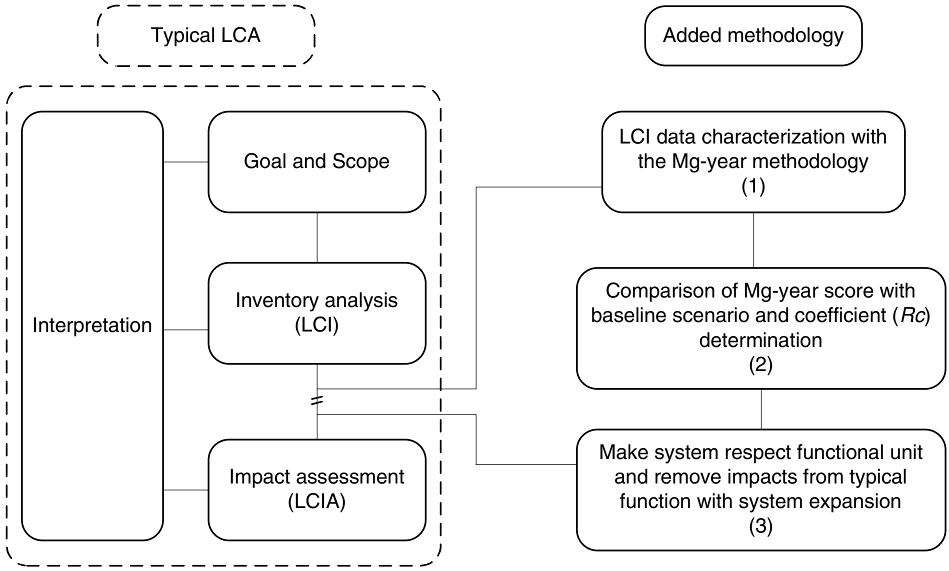

Figure 2 Life cycle assessment (LCA) framework for carbon mitigation project analysis (adapted from ISO 2006a).

forcing of the baseline and the carbon mitigation scenarios calculated in the previous step, and q_n is the Mg-year reduction achieved in n years chosen as the functional unit. Essentially, Rc corresponds to the reference flow required for the carbon mitigation project to achieve the desired Mg-year reduction. For the biofuel example, it corresponds to the kilometers needed for a biofuel scenario that replaces a petroleum scenario to achieve the Mg-year reduction.

3. The last step of the methodology is to recalculate the inventory analysis (or impacts) so that it respects the Mg-year reduction functional unit. One does this by removing impacts related to the typical function (usually the baseline scenario) through system expansion and by multiplying with the reduction coefficient. This can be done at the LCI stage or at the impact assessment stage with equation (6):

$$\begin{bmatrix} I_{m1} \\ \vdots \\ I_{mi} \end{bmatrix} = Rc \left[\begin{bmatrix} I_{cm1} \\ \vdots \\ I_{cmi} \end{bmatrix} - \begin{bmatrix} I_{b1} \\ \vdots \\ I_{bi} \end{bmatrix} \right] \quad (6)$$

where I_{mi} represents the inventory (or impact score) of category i related to the mitigation function, Rc is the reduction coefficient, and I_{cm} and I_b are the inventory (or impact score) generated by the carbon mitigation and baseline scenarios. An exception can be made for the climate change impact category, where I_m equals I_{cm}. Therefore, the climate change impact can be seen as an efficiency measure—in other words, the GHG emission necessary to achieve the Mg-year reduction.

Case Study

Goal and Scope

In the following sections, the presented framework is applied to a biofuel case study. Steps are presented as in a traditional LCA.

The biofuels studied are maize ethanol produced in the United States, sugar cane ethanol produced in Brazil, and cellulosic ethanol made from willow plantation in the United States. These scenarios were chosen because high-quality data concerning their processes are available. Corn ethanol and sugarcane ethanol represent 89% of the present ethanol production

(Renewable Fuels Association 2008). We included willow plantation ethanol in the study to allow a comparison with a second-generation pathway. A petrol Euro3 scenario is used as the baseline scenario.

The system boundaries include cultivation, transportation of feedstock to refinery, the refinery process, transportation to Europe (by oceanic tankers), distribution of ethanol, and ethanol-fuelled vehicle operation. To evaluate potential GHG savings from biofuel use, we also include in the system boundary a conventional vehicle operation respecting Euro3 standards. The environmental burdens associated with the vehicle and with the road system and its maintenance are not included in the analysis, as they are not relevant to this study.

The corn ethanol scenario includes cultivation of maize, including pesticides, herbicides, fertilizers, consequent emissions of N_2O and ammonia (NH_3), and tractor operation. The corn is transported and transformed in a dry mill. The data correspond to the average U.S. context.

The sugarcane pathway is modeled as sugarcane cultivation with 20% mechanical and 80% manual harvest. The ethanol is produced in fermentation plants (83%) and sugar refineries (17%). Coproducts, considered through economic allocation, include electricity from bagasse for the fermentation plant and sugar, bagasse, vinasse, and electricity from bagasse from the sugar refinery.

Willow plantations are cultivated in a short-rotation intensive culture. Processes include chemical product and relevant emissions, cultivation, bailing harvest, and storage. Wood is chipped at the cellulosic fermentation plant.

The anhydrous ethanol, transported on transoceanic tankers to Europe, is burned in a flex-fuel vehicle in an E85 blend. The emissions are normalized to match the energy efficiency of a petrol Euro3 engine. IMPACT2002+ was chosen to assess the environmental impacts of both LCAs as it gives aggregated end-point categories, including climate change impact. The focus is not on the results themselves but on the effect of using the suggested framework. Results are presented in four damage categories: human health in disability-adjusted life years (DALYs), ecosystem quality in potentially disappeared fraction of species integrated over surface area and time (PDF*m^2*yr), climate change in kg CO_2eq, and resources in MJ primary energy.[3] To characterize the biodiversity impacts of LUC, we integrate the characterization factors related to land transformation of Eco-Indicator99 into the ecosystem quality category of Impact2002+. This is possible because Impact2002+ is partially based on Eco-Indication99 and end-point categories are compatible.

Transportation LCA

In this study, the function is defined as driving a passenger vehicle. The functional unit is driving the light-duty vehicle for 1 km.

Carbon Mitigation LCA

The carbon mitigation projects studied have a functional unit of 1 Mg-year reduction compared to a baseline scenario achieved after 50 years. A 50-year time frame for the stabilization of GHG emissions corresponds to the category IV set of scenarios analyzed by the IPCC. This category has the most assessed scenarios and models a temperature increase of 3.2°C to 4°C. Therefore, any scenario that cannot yield any "climate" benefit within the 50-year time frame is rejected. The IPCC category IV set of scenarios corresponds to the worst acceptable scenarios (IPCC 2007a). Also, as most carbon mitigation projects have a constant benefit (carbon avoidance) over time, comparing different projects over 50 or 100 years would not yield significant differences. Finally, as the carbon benefit of most carbon mitigation projects is calculated by comparison with a conventional scenario (e.g., biofuel replacing petroleum), the validity of this replacement will decrease with time as technology improves. Therefore, a time horizon longer than 50 years could lead to insignificant results, as true carbon avoidance would not be modeled correctly.

The LUC emissions are integrated in a theoretical approach as a subsequent step. We do not attempt to evaluate future land displacement with market models and the like. The goal here is to understand the added value of using the Mg-year approach to assess LUC emissions and to evaluate the added environmental impacts when land displacement is included in the analysis.

Results concerning payback time are compared with those of other studies. The payback time corresponds to the year in which the CRF of the biofuel scenario, including LUC emissions, equals the CRF of the fossil scenario.

Although many LUCs are possible through indirect allocation (Searchinger et al. 2008), only emissions from direct land displacement are considered. The scenarios studied are replacement of (1) a North American prairie grass land with maize and willow plantations and (2) a Cerrado land with sugarcane crops. These choices are considered to have a higher probability of occurring (Fargione et al. 2008).

Life Cycle Inventory (LCI)

The LCI of the previously described scenarios is computed with data mostly taken from the ecoinvent 2.0 database. The willow plantation process is taken from the biomass-to-liquid (BtL) study (Jungbluth et al. 2008) and tailpipe emissions of E85 combustion from Graham and colleagues (2008). Coproducts are allocated economically. Simapro 7.1 was used for the LCI modeling.

Considerable uncertainties exist concerning the evaluation of LUC emissions because of the heterogeneity of biomass, different means of land clearing, and uncertainty in burning efficiency. Corresponding data were collected from the literature (Fearnside 1997; Piñeiro et al. 2009) and represent average emissions for average Cerrado and American grassland. LUC activities cause large amounts of emissions during the clearing year but also emit during following years through biomass decay. Because of lack of data and high uncertainty, we consider GHG released from LUC as emitted totally in the 1st year. Also, all emissions caused by LUC are attributed to the biofuel scenario; in other words, no allocation is done among other possible coproducts generated by the crop.

Results and Discussion

Applying the Methodology

GHG Inventory in Lashof (1)

The GHG-related emissions of the biofuel pathways and petrol pathway are compiled with the Lashof method for 1 km driven each year for 50 years. For example, the GHG inventory from the LCI of the maize scenario and corresponding CRF are presented in table 1. These four gases were chosen because they represent 99.8% of a 100-year GWP score. Figure 3a shows the progression of the Mg-year score of the studied scenario. As time advances, the Mg-year score increases faster because anterior emissions accumulate radiative forcing.

Table 1 Life cycle inventory of GHGs and CRF for 1 kilometer driven with maize-based E85

GHG	CO_2	*CO*	CH_4	N_2O	*CRF (Mg-year)*
g emitted	166	0.199	0.241	0.186	0.187

Note: One gram (g) = 10^{-3} kilograms (kg, SI) ≈ 0.035 ounces (oz). GHG = greenhouse gas; CRF = cumulative radiative forcing; CO_2 = carbon dioxide; CO = carbon monoxide; CH_4 = methane; N_2O = dinitrogen oxide; Mg-year = megagram-year.

Coefficient Determination (2)

With the Mg-year score calculated for the petroleum and the biofuel scenarios, it is possible to calculate the reduction coefficient (*Rc*) with equation (5). As the functional unit is 1 Mg-year reduction over 50 years, q_n is equal to 1 Mg-year, and CRF_b and CRF_{cm} are the cumulative radiative forcing of the petroleum scenario and the biofuel scenario, respectively. The *Rc* value for the maize scenario is 125, which means that replacing 125 km of fossil transport with corn-based E85-fuelled transport each year for 50 years will avoid 1 Mg-year of radiative forcing. The *Rc* values for sugarcane and willow plantation scenarios are 7.4 and 13, respectively.

Functional Unit and System Expansion (3)

Biofuels are viewed as a multioutput product that delivers a transportation function and a carbon mitigation function. To isolate the impacts of the mitigation function of the biofuel from the transportation function, we subtracted impacts from the baseline scenario (petrol Euro3), considering that all biofuel products replace equivalent amounts of petrol. This hypothesis neglects the fact that the introduction of biofuel in the market could result in an increase in fuel demand,

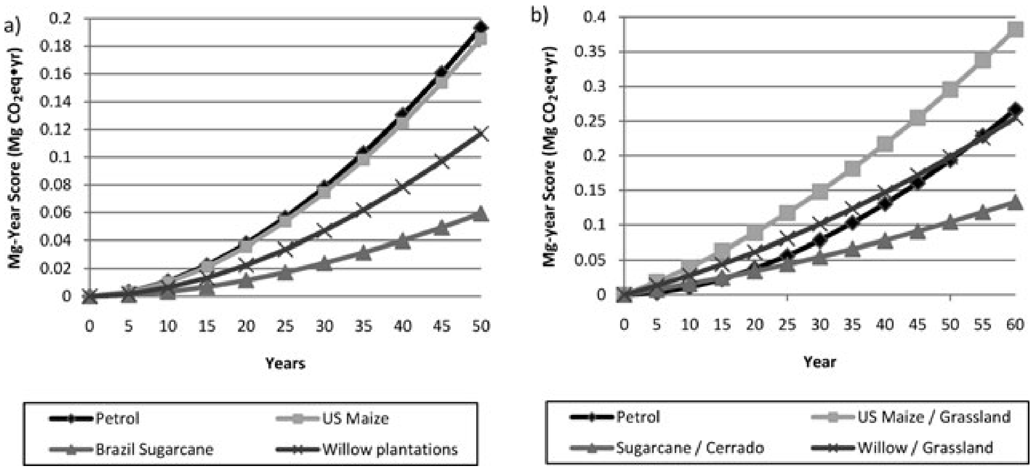

Figure 3 (a) Megagram-year (Mg-year) score of studied biofuels for 1 kilometer driven each year for 50 years, and (b) Mg-year score of studied biofuel for 1 kilometer driven each year for 50 years, including land-use change emissions.

thus invalidating the total fuel replacement hypothesis.

Life Cycle Impact Assessment

Results, relative to each category's highest impact, are shown in figure 4 for the typical transport LCA (see figure 4a) and the carbon mitigation LCA (see figure 4b). For the typical LCA, all three biofuel pathways have a lower impact in climate change than the fossil scenario, with scores of 82%, 31%, and 55% for corn, sugarcane, and willow scenarios, respectively. IMPACT2002+ uses the 500-year GWP. When we use the Mg-year accounting for a 50-year project (see figure 3a), the reductions correspond to 96%, 31%, and 60% of the Mg-year accumulated by the fossil scenario. These higher results are caused by the different weight attributed to N_2O emissions. The Lashof method is equivalent to attributing a GWP of 50 years for the 1st year of emissions, a 49-year GWP for the 2nd year of emissions, and so on. Hence, the GWP mean time frame for N_2O emissions is shorter than 50 years. The instant radiative efficiency of N_2O is 216 times higher than that of CO_2, and decaying is relatively slow (half-life of 114 years), whereas the 500-year GWP has a value of 156 for N_2O.

The results of figure 4a are similar to those of a previously published study (Zah et al. 2007), as the two use mostly the same inventory data. The high impact of the sugarcane scenario for human health is caused by the use of pesticides and herbicides, such as arsenic and Aldrin. The negative score of willow plantation in ecosystem quality comes from the phytoremediation properties of this crop.

For the carbon mitigation LCA (see figure 4b), the human health impact is dominated by sugarcane ethanol, as in the typical LCA, but the ecosystem quality is now dominated by the corn ethanol pathway. It is impossible to determine the best scenario, as no biofuel is best in all categories. The resource utilization category is negative for all biofuel pathways, led by corn ethanol because it is the scenario that would displace the most kilometers of fossil-based transport. The climate change category results represent the GHGs emitted to fulfill the functional unit, characterized with the 500-year GWP. For example, in the maize scenario, this category represents the GHGs emitted when one drives 125 km. This category is dominated by the corn ethanol scenario, once again because of its high number of kilometers needed to fulfill the functional unit.

The choice of the baseline scenario and the consequent efficiency at reducing carbon

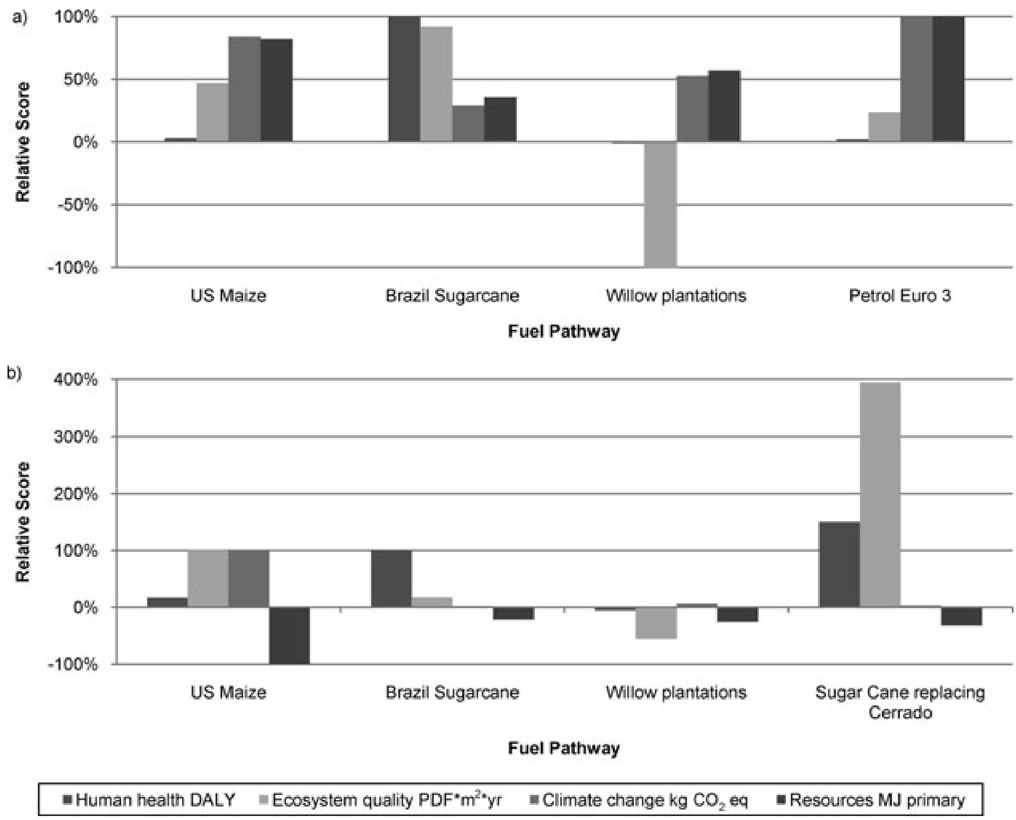

Figure 4 (a) End-point damage from a transportation point of view for biofuel and baseline scenarios, and (b) end-point damage from a carbon mitigation point of view for biofuel with and without land-use change emissions included. DALY = disability-adjusted life year; PDF*m^2*yr = potentially disappeared fraction of species integrated over surface area and time; kg CO_2eq = kilograms of carbon dioxide equivalent; MJ = megajoules.

emissions of the biofuel pathway are critical factors that have a high impact on the carbon mitigation LCA results. For the system expansion step, a high *Rc* is advantageous when the impacts of the studied scenario are lower than those of the baseline scenario but disadvantageous when impacts are higher. Comparing, for example, the Brazil sugarcane pathway and the U.S. maize pathway demonstrates this. The first has a high carbon-reducing efficiency and thus a low *Rc*, whereas the opposite is true for the second. Both pathways have a higher impact than the petrol scenario in ecosystem quality and a lower impact in resource utilization. The low *Rc* of the Brazil sugarcane scenario is advantageous in the ecosystem quality category, as it brings this impact from worse than U.S. maize in the transport LCA to better than maize in the carbon mitigation LCA. The opposite is true in the resource utilization category, where the high *Rc* of U.S. maize makes the worst scenario of this impact category become the best. But because in this study the resource utilization category is mainly related to fossil energy, it is possible to say that the U.S. maize scenario needs to avoid high amounts of energy use to achieve the 1 Mg-year reduction. The negative results of the carbon mitigation LCA do not always guarantee an environmental gain. Thus, one must take special care when choosing a baseline scenario and analyzing the impacts.

The approach taken to assess the climate change impact can be called dynamic because it accounts for emissions year by year and therefore gives the impact on a yearly basis for climate

Table 2 Mg-year score of different types of land-use change (LUC) emissions per hectare

	Land type		
	Tropical forest	*Cerrado*	*American grassland*
Mg-year/ha	21,379	2,460	3,772

Note: One hectare (ha) = 0.01 square kilometers (km^2, SI) ≈ 0.00386 square miles ≈ 2.47 acres. Mg-year/ha = megagram-year per hectare.

change and GHGs. To have the best evaluation possible, one could address a certain time period of emission for all GHGs, such as those caused by infrastructure building. Unfortunately, present databases do not easily allow for this kind of dynamic inventory.

LUC Emission

Inventory results indicate that 0.29, 0.18, and 0.21 m^2 of cultivated land are necessary to fulfill the 1 km function of the maize, sugarcane, and willow plantation scenarios, respectively. Table 2 presents Mg-year scores representing the 50-year CRF generated by the LUC emission of 1 hectare of converted land. The tropical forest impact is presented for comparison purposes only.

When LUC emissions are included, the Mg-year score will be greatly affected, as the GHG emitted in the 1st year will accumulate important radiative forcing. As figure 3b shows, only the Brazil sugarcane scenario can be considered in the carbon mitigation LCA, as it is the only one that achieves a lower CRF than the baseline project in less than 50 years. Figure 4b presents a life cycle impact assessment of the sugarcane scenario as a carbon mitigation project, including impact from Cerrado land transformation. The important difference with the scenario that excludes LUC is the reduction coefficient (*Rc*), which is now 11.2 instead of 7.4. This means that one needs to drive 50% more kilometers every year to achieve the 1 Mg-year reduction. Furthermore, the ecological impact that originates from the land transformation is high, representing 91% of the ecosystem quality category. This is due to a high biodiversity value of the Cerrado and a low value for the cultivated land.

Payback times are equal to 332, 19, and 55 years for the maize, sugarcane, and willow plantation scenarios, respectively. When we use the Lashof methodology to calculate the payback time of a given project, the results are significantly higher than in other studies (Fearnside 1997; Piñeiro et al. 2009). This is because LUC emissions accumulate radiative forcing at the beginning of the studied time frame, whereas other studies that include LUC emissions consider emissions made at the beginning of the study as equivalent to those emitted near the end. For example, Fargione et al. (2008) calculated a payback time of 93 years for the maize ethanol displacing prairie grass. When we use the Lashof method, payback time is 332 years.

Including LUC emission in biofuel LCA has always been a challenge, because the emissions are amortized over a subjective time frame. When we investigate biofuel environmental impacts from a carbon mitigation perspective, the LUC emissions are integrated in the functional unit.

Conclusions

As more and more carbon mitigation projects are initiated, a holistic methodology, such as LCA, is necessary to help decision makers compare available options. In this study, we have successfully applied a novel framework to a biofuel case study, which shows that the proposed framework enables the comparison of different carbon mitigation projects by comparing them on a CRF reduction basis through Mg-year accounting. The advantage of the Mg-year methodology is the temporal characterization that weighs GHG emissions made now and those emitted in the future differently. Although LCA is privileged in the proposed tool, including the Lashof accounting methodology with other assessment tools, such as carbon footprinting, could be beneficial, especially in analysis of carbon mitigation measures.

The limitations of the framework are closely related to the quality of the data used. For Mg-year scores, uncertainty comes from the data concerning GHG decay and subsequent formulas, which will inevitably vary in the future as the atmospheric carbon concentration rises. Further uncertainties also arise from limited modeling technology for long periods of time, as the efficiency of the studied processes will improve instead of accrue.

We believe the framework can assess any kind of project, as they can all be characterized with Mg-year accounting. Future research should aim at broadening the use of the framework to other types of carbon mitigation projects to verify its applicability. Work should also be done to enlarge the inventory data of the carbon mitigation project. The modeling of biotic sequestration needs more development. For example, one would need to characterize the benefits of transforming a cropped land to a forest or to assess environmental consequences of ocean fertilization for algae growth.

Acknowledgement

We thank the industrial partners of the International Chair in LCA for their support.

Notes

1. One megagram (Mg) = 1 tonne (t) = 10^3 kilograms (kg, SI) ≈ 1.102 short tons.
2. One kilometer (km, SI) ≈ 0.621 miles (mi).
3. One kilogram (kg, SI) ≈ 2.204 pounds (lb); one megajoule (MJ) = 10^6 joules (J, SI) ≈ 239 kilocalories (kcal) ≈ 948 British Thermal Units (BTU).

References

Fargione, J., J. Hill, D. Tilman, S. Polasky, and P. Hawthorne. 2008. Land clearing and the biofuel carbon debt. *Science* 319(5867): 1235–1238.

Fearnside, P. M. 1997. Greenhouse gases from deforestation in Brazilian Amazonia: Net committed emissions. *Climatic Change* 35(3): 321–360.

Fearnside, P. M. 2000. Uncertainty in land-use change and forestry sector mitigation options for global warming: Plantation silviculture versus avoided deforestation. *Biomass and Bioenergy* 18(6): 457–468.

Fearnside, P. M. 2002. Why a 100-year time horizon should be used for global warming mitigation calculations. *Mitigation and Adaptation Strategies for Global Change* 7(1): 19–30.

Fearnside, P. M., D. A. Lashof, and P. Moura-Costa. 2000. Accounting for time in mitigating global warming through land-use change and forestry. *Mitigation and Adaptation Strategies for Global Change* 5(3): 239–270.

Fu, G., A. Chan, and D. Minns. 2003. Life cycle assessment of bio-ethanol derived from cellulose. *International Journal of Life Cycle Assessment* 8(3): 137–141.

Graham, L. A., S. L. Belisle, and C.-L. Baas. 2008. Emissions from light duty gasoline vehicles operating on low blend ethanol gasoline and E85. *Atmospheric Environment* 42(19): 4498–4516.

IPCC. 2000. *Land use, land-use change and forestry*. Cambridge, UK: Cambridge University Press.

IPCC. 2007a. *Climate change 2007: Synthesis report*. Valencia, Spain: IPCC.

IPCC. 2007b. *Climate change 2007: The physical science basis*. Valencia, Spain: IPCC.

ISO. 2006a. *ISO 14040: Environmental management—life cycle assessment—principles and framework*. Geneva, Switzerland: ISO.

ISO. 2006b. *ISO 14044: Environmental management—life cycle assessment—requirements and guidelines*. Geneva, Switzerland: ISO.

Joos, F., I. C. Prentice, S. Sitch, R. Meyer, G. Hooss, G.-K. Plattner, S. Gerber, and K. Hasselmann. 2001. Global warming feedbacks on terrestrial carbon uptake under the Intergovernmental Panel on Climate Change (IPCC) emission scenarios. *Global Biogeochemical Cycles* 15/4: 891–907.

Jungbluth, N., S. Büsser, R. Frischknecht, and M. Tuchschmid. 2008. *Life cycle assessment of biomass-to-liquid fuels*. Bern, Switzerland: ESU-Services Ltd.

Khoo, H. H. and R. B. H. Tan. 2006a. Life cycle evaluation of CO_2 recovery and mineral sequestration alternatives. *Environmental Progress* 25(3): 208–217.

Khoo, H. H. and R. B. H. Tan. 2006b. Life cycle investigation of CO_2 recovery and sequestration. *Environmental Science & Technology* 40(12): 4016–4024.

Kim, S. and B. Dale. 2006. Ethanol fuels: E10 or E85—life cycle perspectives. *International Journal of Life Cycle Assessment* 11(2): 117–121.

Kirschbaum, M. U. F. 2003. To sink or burn? A discussion of the potential contributions of forests to greenhouse gas balances through storing carbon or providing biofuels. *Biomass and Bioenergy* 24(4–5): 297–310.

Korhonen, R., K. Pingoud, I. Savolainen, and R. Matthews. 2002. The role of carbon sequestration and the tonne-year approach in fulfilling the objective of climate convention. *Environmental Science & Policy* 5(6): 429–441.

Lal, R. 2008. Carbon sequestration. *Philosophical Transactions of the Royal Society B: Biological Sciences* 363(1492): 815–830.

Moura Costa, P. and C. Wilson. 2000. An equivalence factor between CO_2 avoided emissions and sequestration: Description and applications in forestry. *Mitigation and Adaptation Strategies for Global Change* 5(1): 51–60.

Piñeiro, G., E. G. Jobbágy, J. Baker, B. C. Murray, and R. B. Jackson. 2009. Set-asides can be better climate investment than corn ethanol. *Ecological Applications* 19(2): 277–282.

Renewable Fuels Association. 2008. World fuel ethanol production. www.ethanolrfa.org/industry/statistics/. Accessed 7 April 2009.

Searchinger, T., R. Heimlich, R. A. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, D. Hayes, and T.-H. Yu. 2008. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. *Science* 319(5867): 1238–1240.

Sheehan, J., A. Aden, K. Paustian, K. Killian, J. Brenner, M. Walsh, and R. Nelson. 2003. Energy and environmental aspects of using corn stover for fuel ethanol. *Journal of Industrial Ecology* 7(3–4): 117–146.

Zah, R., H. Böni, M. Gauch, R. Hischier, M. Lehmann, and P. Wäger. 2007. *Life cycle assessment of energy products: Environmental assessment of biofuels*. Bern, Switzerland: Empa Technology and Society Lab.

About the Authors

Alexandre Courchesne and **Valérie Bécaert** are research associates for the CIRAIG at the École Polytechnique de Montréal, Canada. **Ralph K. Rosenbaum** was a research officer and lecturer for the CIRAIG at the École Polytechnique de Montréal at the time the article was written and is currently associate professor at the Section for Quantitative Sustainability Assessment in the Department of Management Engineering at the Technical University of Denmark. **Louise Deschênes** and **Réjean Samson** are professors for the CIRAIG at the École Polytechnique de Montréal.