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Analysis

The land use change time-accounting failure

Marion Dupoux*

Department of Economics, University of Gothenburg, Vasagatan 1, PO Box 640, 405 30 Göteborg, Sweden
IFP Energies Nouvelles, Rueil-Malmaison, France
University of Paris Nanterre, Nanterre, France
Economie Publique, AgroParisTech, INRA, University of Paris-Saclay, Thiverval-Grignon, France

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ABSTRACT

This paper builds on the disconnection between scientific evidence and policy assumptions about the temporal profile of land use change (LUC) emissions. Whereas natural scientists find evidence of a decreasing time profile of LUC emissions, European energy policy relies on a steady time profile. We investigate the consequences of using such a uniform (constant) time profile when assessing biofuel projects with cost-benefit analysis, a widespread economic tool for public project assessment. We show that the use of the uniform time profile distorts LUC emissions costs downwards (upwards) when carbon prices grow slower (faster) than the discount rate. We illustrate our results with the conversion of grassland to wheat cultivation for bioethanol production in France. Under current assumptions in public project assessment, we find a 70% overestimation of costs related to direct LUC emissions. We propose two tools to aid in decision-making and address the decision error. Finally, we provide contextual policy recommendations.

1. Introduction

While biofuels were originally considered an important tool in the response to global warming, their sustainability has been questioned since the study by Searchinger et al. (2008). This study pointed out that land use change (LUC) emissions could partly or even totally cancel out the environmental benefits of using biofuels instead of fossil fuels. Consequently, LUC impacts have taken more and more space in European energy and environmental policies (European Commission, 2015a, 2018a, b). LUC emissions resulting from the conversion of land with high carbon concentrations (e.g. grassland and forestland) to land with low carbon concentrations (e.g. cropland)[1] are unique in their distribution over time as they do not follow a steady time profile in the same way industrial emissions do (Broch et al., 2013). Instead, LUC emissions are mostly immediate (Guo and Gifford, 2002; Murty et al., 2002; Zinn et al., 2005; De Gorter and Tsur, 2010; Delucchi, 2011; Searchinger et al., 2018). Land conversion to energy crop farming causes a disturbance that translates into carbon stock changes and in turn carbon emissions. The disturbance is twofold and spreads over time differently (e.g. Marshall, 2009; Delucchi, 2011): emissions are (*i*) roughly *immediate* when related to above- and below-ground biomass and (*ii*) *decreasing* over a longer time period when related to soil (Poeplau et al., 2011). More particularly in temperate regions, which is are our focus in this paper, scientists have found that carbon releases from soils following conversion of grassland or forestland to cropland decrease exponentially over time (see the meta-analysis by Poeplau et al. (2011)). Such a temporal profile has been consistently referred to in later studies (e.g. Nyawira et al., 2016; Li et al., 2018; Searchinger et al., 2018).

In this paper, we investigate the disconnection between scientific and policy considerations of the temporal profile of LUC emissions. Indeed, European policies assume that LUC emissions, irrespective of type of carbon sink, have a *uniform* (constant) time profile (European Commission, 2009a; European Commission, 2015b; European Commission, 2018a). What are the consequences of such an assumption on the assessment of biofuel-related investment projects? Shedding

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* Department of Economics, University of Gothenburg, Vasagatan 1, PO Box 640, 405 30 Göteborg, Sweden.
E-mail address: marion.dupoux@economics.gu.se.

[1] This type of conversion, often related to first-generation biofuels, is the main focus of our paper. By contrast, second-generation biofuels, related to other types of biomass such as perennial grasses, may store more carbon than previous land use such as annual cropland. Sequestrations will not be numerically investigated since our research is primarily related to emissions caused by biofuels. Nonetheless, sequestrations are discussed in Section 5.

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light on this question and suggesting tools to support decision-making in this context are the two main objectives of the present paper.

The *ex ante* assessment of projects relies on a variety of approaches, e.g. multicriteria analysis, cost-benefit analysis (CBA), risk assessment and public participation, that complement each other to support the decision of whether or not a project should be implemented. In practice, CBA is a widely used tool in the assessment of public investment projects in the energy and transport sectors (OECD, 2018b).[2] It is reported that the influence of CBA on the decision of whether to implement a project is moderate to large (ibid). Discount rate and time path of carbon prices are the two key elements of CBA. Both affect emissions at different times differently except when carbon prices grow exactly at the discount rate, i.e. when the Hotelling rule applies. This rule prevents the discounting effect from overwhelming the value of emissions over time and is widespread in climate change modelling (e.g. Dietz and Fankhauser, 2010) and the determination of shadow carbon values (e.g. Quinet, 2009; Quinet, 2019, in the long term). Nonetheless, in current practice, carbon prices usually deviate from this rule (Hoel, 2009), at least temporarily. This is because they need to reflect increasingly stringent objectives to curb greenhouse gas (GHG) emissions (for example, the goal of limiting the increase in average global temperature by 2 °C became a goal of limiting the increase to 1.5 °C (Rogelj et al., 2018, IPCC report)). This requires a progressive alignment with the Hotelling rule from current, relatively low, carbon prices.

We develop a two-period model to show that the use of a uniform distribution of LUC impacts over time associated with the common deviation of carbon prices from the Hotelling rule leads to a distortion of the net present values (NPVs) of projects. We compute the net present values of LUC-related emissions under the two different time distributions of LUC emissions: the *uniform* (constant emissions) time distribution typically, yet wrongly, assumed in European policy, and the *differentiated* (across time) distribution, which reflects the proper dynamics of emissions after land conversion, as put forward by natural scientists. We find that, if the carbon price increases slower (faster) than the discount rate, the costs of LUC emissions are underestimated (overestimated) under the uniform approach compared with under the differentiated approach that reflects biophysical reality.

We illustrate our results with the case of French bioethanol production from wheat. Because of the complexity of the quantification of indirect LUC (see e.g. Di Lucia et al., 2012), we focus on direct LUC,[3] which accounts for approximately half of LUC emissions associated with wheat-based ethanol (Fritsche et al., 2010). Under the assumptions used in France for project assessment, i.e. a 4.5% discount rate (Quinet, 2013; France Stratégie, 2017) and the shadow price of carbon estimated in Quinet (2019, p.32), we find that the LUC-related NPV of a bioethanol project that entails a conversion of grassland into cropland[4] is underestimated by almost 70%. We explore more carbon price scenarios and find that the misestimation of the value of LUC emissions ranges from −70% to +23%.

With the current practice of CBA in project appraisal (OECD, 2018b) and the current use of uniform time distribution (European Commission, 2018a), the challenge is to provide guidelines for decision-makers when faced with biofuel projects. CBA should certainly not be the only tool supporting decision-making (Norgaard, 1989). Nonetheless, as CBA is reported as influential in decision-making (OECD, 2018b), it should be used properly to support decisions. Therefore, we provide two convenient tools to support decision-making in this context. The first tool is the compensatory rate, which cancels out the value difference between the uniform and the differentiated time profile. This rate is useful in that it can be compared with the discount rate chosen for the project evaluation, and this comparison can in turn indicate in which direction decision-makers misestimate the LUC costs. The second tool is the carbon profitability (CP) payback period. Contrary to the classical carbon payback period stemming from the (physical) carbon debt concept, the CP payback period is price-based and likely to better incentivise reductions of LUC emissions. We recommend the use of a CP payback period benchmark predetermined by policy-makers for the purpose of comparing the uniform and differentiated approaches. These two tools are provided in a Python program available online, namely PyLUCCBA, described in this paper's supplementary material.[5] PyLUCCBA computes NPVs of LUC emissions under both the uniform approach (mimicking the European energy policy method) and the differentiated approach (based on the meta-analysis of Poeplau et al. (2011)) as well as project-specific non-LUC emissions.

The paper is organised as follows. Section 2 reviews the literature on the particular time distribution of LUC emissions and compares it with the assumption of constant emissions over time employed by the European Commission in the context of project assessment. Section 3 presents the theoretical model and derives the impacts of using the uniform time distribution on the NPV of a project. These results are applied to the French production of wheat-based ethanol, leading to a quantification of the distortion of LUC emissions costs under the uniform approach. Section 4 proposes two simple tools created to aid in decision-making regarding biofuel-related projects expected to affect global warming. Section 5 discusses the assumptions of our model, the implications of our results for indirect LUC and projects entailing carbon sequestrations, and finally the implications of the discrepancy between temporal distributions in the context of carbon markets. Section 6 concludes the paper and provides policy recommendations.

2. Background

In 2009, the European Commission imposed a mandatory goal for member states to ensure a 10% minimum share of renewable energies (and particularly a 6% share of biofuels[6]) in transport petrol and diesel by 2020 (European Commission, 2009a, Renewable Energy Directive (RED)). Although the sustainability criteria of biofuels mentioned that the whole life cycle of biofuels must be considered when assessed (European Commission, 2009b), the study by Searchinger et al. (2008) pointed out the LUC issue and the extent to which it might result in a worse carbon balance for biomass-based fuels compared with that for fossil-based fuels. As LUC became critical to the determination of the carbon balance of biofuels (Fargione et al., 2008), it led policy-makers to amend the 2009 RED in order to include the indirect LUC impacts that biofuel projects might cause (European Commission, 2015b). In this section, we review the literature on the dynamics of LUC as estimated in scientific literature (Section 2.1) and as assumed in European energy policies (Section 2.2). We then raise the issue of the discrepancy between these two ways of accounting for LUC dynamics when it comes to the assessment of public investment projects (Section 2.3).

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[2] This report relies on a questionnaire addressed to OECD countries about their current use of cost-benefit analysis in project assessment. Carbon values as well as discount rates used in each country are provided along with the extent to which CBA is used and influential in decision-making.

[3] *Direct* LUC refers to the replacement of a given land with cropland entirely dedicated to biofuel production. *Indirect* LUC occurs when the replacement of land dedicated to food crops with farming of biofuel crops reduces the availability of land for food production. This reduction may be compensated for in other places where land is converted to use for food crops, thereby potentially generating carbon emissions. Indirect LUC is more difficult to quantify because it involves economic forces (see e.g. Feng and Babcock, 2010) following an increased production of biofuels and therefore often requires modelling. Nevertheless, the mechanism at the origin of LUC emissions is the same for both categories of LUC. We extend the discussion of our results to indirect LUC in Section 5.

[4] Such land conversion is responsible for most LUC emissions related to the production of biofuels in France (Chakir and Vermont, 2013).

[5] The tool is available on GitHub, *https://github.com/lfaucheux/PyLUCCBA* and on Pypi, https://pypi.org/project/PyLUCCBA/.

[6] Then increased to 7% by the European Commission (2015b).

2.1. LUC emissions temporal profile in academic research

Land conversion results in carbon stock changes. The carbon balance disturbance occurs in biomass and soil, both of which constitute important carbon sinks.[7] Depending on the carbon concentration in both the initial and the final land, land conversion can either release carbon, generating CO_2 emissions to the atmosphere, or store carbon, leading to CO_2 sequestrations from the atmosphere. The present paper tackles the issue of emissions but extends the discussion to sequestrations in Section 5. The dynamics of carbon losses are sink-specific. While the change in biomass carbon stock is in most cases instantaneous (Delucchi, 2011), changes in soil organic carbon (SOC) stock occur over the course of several[8] years until the carbon stock reaches a new equilibrium (Marshall, 2009; De Gorter and Tsur, 2010; Delucchi, 2011; Poeplau et al., 2011; Don et al., 2012). Measuring SOC is a complex task (Anderson-Teixeira et al., 2009). Nonetheless, there is a large literature on the dynamics of SOC changes due to LUC. Some *assume* certain carbon response functions, such as linear (e.g., Anderson-Teixeira et al., 2009) or exponential (e.g., Evrendilek et al., 2004; Delucchi, 2011) SOC stock losses over time. Others investigate the carbon response function that best fits SOC stock changes for different land conversions by means of meta-analyses (e.g. Poeplau et al., 2011; Fujisaki et al., 2015, in the context of temperate and tropical regions, respectively). The carbon response functions developed by Poeplau et al. (2011), based on empirical data, have often been considered a reference for temperate zones in later studies (e.g., Nyawira et al., 2016; Li et al., 2018; Searchinger et al., 2018). In particular, the conversion of both grassland and forestland to cropland is characterised by an exponential decrease in SOC stocks.[9] Overall, empirical evidence suggests that, when a land accumulates and maintains carbon stocks better than another land, the conversion of the former to the latter results in carbon losses that tend to decrease over time.

2.2. LUC emissions temporal profile in EU policy

As much in the 2009 RED as in the more recent 2018 RED, LUC emissions are assumed to be uniformly distributed across time: “[a]nnualised emissions from carbon stock changes caused by land-use change […] shall be calculated by dividing total emissions equally over 20 years” (European Commission, 2009a; European Commission, 2018a). In other words, LUC emissions are summed over the 20-year time horizon and divided evenly across years. While such a uniform temporal profile holds for emissions generated from the cultivation of energy crops (e.g. yearly input and tillage practices) and biofuel production (e.g. emissions due to the yearly production process, transport and distribution), it is not suitable for LUC emissions since land conversion occurs just once as a shock. This widespread straight line amortisation method has the advantage of being simple and consistent (Broch and Hoekman, 2012), unfortunately at the expense of not considering the genuine dynamics of LUC emissions. For the sake of clarity, we name the two temporal distributions tackled in this paper as follows:

- Uniform temporal profile: constant time profile as assumed in European energy policies and described in this subsection.

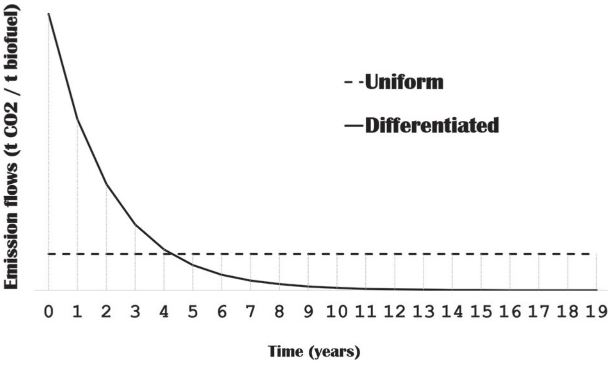

Fig. 1. Temporal profiles of LUC emissions (uniform vs. differentiated).

- Differentiated temporal profile[10]: decreasing time profile as reported in the biophysics literature (see Section 2.1).

These two temporal profiles are illustrated in Fig. 1, where land conversion occurs at time $t = 0$. Note that the sum of emissions under both temporal profiles is the same over the time horizon. Only the dynamics over time varies. The next section sheds light on the issue that may arise from the discrepancy between the two temporal profiles when it comes to project evaluation.

2.3. Why the policy's disconnection from science matters in project assessment

Project assessment relies on a variety of complementary tools such as CBA, multicriteria analysis and risk assessment (OECD, 2018a). Investments in the energy sector are often informed by cost-benefit analyses that include GHG emissions (OECD, 2018b). In France, which is the country for our case study in Section 3.2, socio-economic analysis is even mandatory (France Stratégie, 2017; Quinet, 2019, Box 10 p.139). In practice, final decisions are moderately or largely influenced by CBA results as reported in OECD (2018b, Figure 16.15). All these elements make CBA of biofuel projects worth regarding, especially when projects entail particular temporal dynamics like those of LUC emissions.

Cost-benefit analysis generally relies on *i*) pricing emissions at each point in time[11] and discounting future emissions costs over time.[12] Both carbon prices and the discount rate affect emissions differently over time.[13] Only when carbon prices grow at the discount rate are emissions costs not affected by the time profile of emissions. This is known as the Hotelling rule, originally established for exhaustible resources.[14] The Hotelling rule guarantees that carbon emissions do not suffer from discounting. Nonetheless, it is rarely the case that the discount rate employed in CBA of public investment projects is equal to the rate at

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[7] Soil organic carbon is one of the largest carbon sinks in the earth system, storing 3.3 and 4.5 times as much carbon as atmospheric and biotic carbon pools, respectively (Lal, 2004).

[8] Generally for 20 years after conversion (IPCC, 2006; European Commission, 2010; Delucchi, 2011; Searchinger et al., 2018).

[9] The exponential profile does not hold for all types of land conversion since carbon stock changes are dependent on a multitude of factors such as climatic variables, land management, vegetation type or soil texture (see Poeplau et al. (2011) and Fujisaki et al. (2015) for an overview in temperate and tropical regions, respectively).

[10] I.e., differentiated across time. For a conversion of grassland to cropland, the time profile tends to decline. However, it is not the case for all types of land conversion. We discuss the case of a conversion of cropland to *Miscanthus* in Section 5.

[11] CO_2 price trajectories are increasing over time to reflect the increase of GHG concentration in the atmosphere and its ensuing global-warming threats over time (De Gorter and Tsur, 2010).

[12] In practice, future environmental costs and benefits are discounted in most countries, including France (OECD, 2018b, Figure 16.10).

[13] Carbon prices (discount rates) tend to increase (decrease) the value of emissions over time.

[14] Applied to global warming, this rule assumes that the capacity of the atmosphere to manage a certain concentration of GHGs is an exhaustible resource. The emissions cap determines the amount of allowed emissions within a given period and this amount depletes over time as one emits GHGs. Consuming the entire amount implies an equivalence between emitting 1 tonne of CO2 today or in a year, which in turn implies that the carbon price should increase at the discount rate.

which carbon prices grow over time (Hoel, 2009; Smith and Braathen, 2015; OECD, 2018b).[15][16] Indeed, while the Hotelling rule is considered a relevant rule in the long term, it is justified to temporarily get away from it to smooth the revalorisation of the climate action, and therefore the trajectory of carbon values over time (Quinet, 2019, p.123). Thus, the problem with using a uniform time profile when emissions are actually decreasing over time, lies not so much in how emissions are quantified over time per se (i.e. in physical terms) as in the discounting and pricing of these emissions over time. With *i*) the incorrect time distribution of LUC emissions used in the European energy policy and *ii*) the common use of CBA as a decision-making tool in the decision-making sphere, we address the issue of project appraisal distortion in the context of emissions induced by LUC.

3. Cost-benefit analysis and the time profile of LUC emissions

In this section, we apply the CBA approach to the two temporal profiles of LUC emissions and determine the direction of the bias (Section 3.1) as well as its magnitude in the case of wheat-based ethanol in France (Section 3.2). Because the dynamics of LUC is our main focus, the model exclusively represents the part of CBA that monetises LUC (carbon-related) impacts.[17][18]

3.1. A two-period NPV model

Consider two periods $t = \{0,1\}$, and denote as $z_t \in \mathbb{R}^+$ the actual emission flow occurring at time *t*. The model aims to compare the LUC-related NPV under the uniform (u) and the differentiated (*d*) time distribution. The differentiated approach preserves the actual emission flows as such (i.e. z_t at time *t*). By contrast, the uniform approach averages emissions over a chosen time period (here 2 years), modifying the actual flows z_0 and z_1 into $\frac{z_0+z_1}{2}\ \forall\ t = \{0,1\}$.

Consider a project that releases emissions as a result of land conversion[19] at $t = 0$. The carbon price grows at the carbon price growth rate denoted $g \in [0,1]$ such that the carbon price at $t = 0$ and $t = 1$ is $p_0 \geq 0$ and $p_1 = p_0(1+g) \geq 0$, respectively. Denoting the discount rate used in the project $r \in [0,1]$, the NPVs associated with the uniform and differentiated approaches are such that, for all $z_0,\ z_1 \in \mathbb{R}^+$:

$$NPV_u = -\left(p_0\frac{z_0+z_1}{2} + p_0\frac{(1+g)}{(1+r)}\frac{z_0+z_1}{2}\right) \tag{1}$$

$$NPV_d = -\left(p_0 z_0 + p_0\frac{(1+g)}{(1+r)}z_1\right). \tag{2}$$

The negative sign indicates that emissions constitute a cost to society. In line with the scientific literature, we assume that $z_0 > z_1$ (e.g. Poeplau et al., 2011; Li et al., 2018).

Considering the differentiated time distribution as the baseline (the one that should be accounted for in policy-making), we assess the bias induced by the use of the uniform time distribution. This amounts to analysing the NPV difference $\Delta NPV = NPV_u - NPV_d$, the sign of which provides information about the downward or upward bias induced by the uniform time distribution. Since the discount rate and carbon prices affect emissions differently over time, we first disentangle one effect from the other before analysing the combined effect.

3.1.1. Discounting effect ($0 < r \leq 1$ and $g = 0$)

To isolate the discounting effect, we assume that $p_1 = p_0 > 0$ and a strictly positive discount rate. The NPV difference is

$$\Delta NPV = \frac{p_0 r(z_0 - z_1)}{2(1+r)} > 0, \tag{3}$$

and deriving the NPV difference with respect to the discount rate gives

$$\frac{\partial \Delta NPV}{\partial r} = \frac{p_0(z_0 - z_1)}{2(1+r)^2} > 0, \tag{4}$$

leading to Proposition 1.[20]

Proposition 1. (**discounting effect**) *Employing the uniform time distribution of LUC emissions increases the discounting effect. As a result, the value of projects entailing such emissions is overestimated,* i.e. *the costs of emissions are underestimated. The higher the discount rate, the larger the bias induced.*

The key difference between the uniform and differentiated time distributions is that emissions mostly occur upfront in the latter. Therefore, in the uniform approach, a greater amount of emissions (at $t = 1$) suffer from the discounting effect, which softens the monetary cost of emissions and thus leads to an underestimation of the costs, compared with the differentiated approach, which fully accounts for the decrease in carbon losses.

3.1.2. Carbon price effect ($0 < g \leq 1$ and $r = 0$)

To isolate the carbon price effect, we assume that $g > 0$ (i.e. $p_1 > p_0$) and a zero discount rate. The NPV difference is

$$\Delta NPV = \frac{1}{2}p_0 g(z_1 - z_0) < 0, \tag{5}$$

and deriving the NPV difference with respect to the carbon price growth rate gives

$$\frac{\partial \Delta NPV}{\partial g} = \frac{1}{2}p_0(z_1 - z_0) < 0, \tag{6}$$

leading to Proposition 2.[21]

Proposition 2. (**carbon price effect**) *Employing the uniform time distribution of LUC emissions increases the carbon price effect. As a result, the value of projects entailing such emissions is underestimated,* i.e. *the costs of emissions are overestimated. The higher the carbon price growth rate, the larger the bias induced.*

Because the carbon price is increasing over time, the earlier the emissions the lower their social cost. In the differentiated approach, emissions mostly occur upfront when the carbon price is lower. By contrast, the uniform approach entails emissions equally spread out over time. Therefore, a greater amount of emissions is priced higher at time $t = 1$. Higher priced emissions, which constitute a higher social cost, lead to an underestimated NPV under the uniform approach.

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[15] See values of both carbon prices and discount rates in different countries in Figures 16.7 and 16.11 respectively in OECD (2018b).

[16] Providing an exhaustive literature review on the discount rate that *should* be considered in CBA and the way carbon prices *should* evolve is beyond the scope of this paper. Rather, we emphasise that in the decision-making sphere, the fact that discount rates differ, in practice, from the rate at which carbon prices rise might be problematic when LUC impacts are involved in CBA of investment projects.

[17] The remaining GHG emissions associated with biofuel production processes and cultivation of energy crops are introduced in the analysis in Section 4.2.

[18] The benchmark of bioethanol projects is conventional fossil fuel production, which does not entail land use change emissions as bioethanol projects do.

[19] From high carbon-concentration land (e.g. forestland and grassland) to lower carbon-concentration land (e.g. cropland).

[20] The proof of Proposition 1 is straightforward: $\Delta NPV > 0$ means that $NPV_u > NPV_d$. The positive derivative of ΔNPV with respect to the discount rate indicates that the difference (overestimation) increases with the discount rate.

[21] The proof of Proposition 2 is straightforward: $\Delta NPV < 0$ means that $NPV_u < NPV_d$. The negative derivative of ΔNPV with respect to the carbon price growth rate indicates that the difference is increasingly negative (i.e. the underestimation is increasing), generating an increasing bias induced by the uniform approach.

3.1.3. Combined effect ($0 < r \leq 1$ and $0 < g \leq 1$)

The use of the uniform time distribution in economic appraisals boosts both the discounting effect (which leads to a reduction of the value of future emissions) and the carbon price effect (which leads to an increase in the value of future emissions). Proposition 3 sheds light on the question of which effect outweighs the other when these effects are combined in CBA (proof in Appendix A.1).

Proposition 3. (**combined effect**) *Under the Hotelling rule, no bias is induced by the uniform approach. When the Hotelling rule does not apply, employing the uniform time distribution in CBA causes an upward (downward) bias of the project value if and only if the carbon price grows slower (faster) than the discount rate.*

When the discounting and carbon price effects perfectly cancel each other out, the uniform and differentiated time distributions are strictly equivalent within CBA (i.e. the same NPV). This means that the construction of the carbon price trajectory follows the Hotelling rule.

When the discounting effect outweighs the carbon price effect (see Proposition 1), using the uniform approach results in an upward bias of the project value. In monetary terms, this means that the cost of emissions is given relatively less weight under the uniform approach, leading to an overestimation of the value of the project. A lower carbon price growth rate than the discount rate may be due to the consideration by decision-makers of the uncertainty about the magnitude of environmental damages and advocates for a strong carbon price signal today to incentivise the reduction of emissions immediately (in line with Stern (2006)).

When the carbon price effect dominates the discounting effect, the uniform approach leads to underestimation of the value of the project (see Proposition 2). Under the uniform approach, carbon emissions 'gain' (monetary) value over time even after discounting, whereas under the differentiated approach, emissions 'benefit' virtually nothing from the price hike since emissions occur mainly upfront. Such a situation where the growth rate of the carbon price is greater than the discount rate is likely to occur when the carbon price path starts at a relatively low level, requiring a strong rise to meet future emissions reductions objectives (rather in line with Nordhaus' idea of a "climate policy ramp"). This case is the most common (OECD, 2018b) as we will see in Section 4.

3.2. Numerical illustration: the case of French wheat-based ethanol

France is the biggest bioethanol producer in Europe (USDA, 2018) and its production mainly relies on wheat (Ademe, I Care and Consult, et al., 2017). In this subsection, we provide a numerical illustration of our theoretical results with the example of direct LUC engendered by wheat-based ethanol production in France. The analysis of direct LUC shows that most emissions related to the cultivation of wheat are due to the conversion of grassland (Chakir and Vermont, 2013, page 48), which will therefore be the focus of our study.[22] Direct LUC related to the conversion of grassland to wheat fields in Europe accounts for approximately 30% of total emissions from life cycle and LUC impacts of bioethanol and approximately half of total LUC emissions, i.e., including indirect LUC (Fritsche et al., 2010, Figures 1 and 2).

3.2.1. Assumptions

France is located in a temperate region where the increasing demand for bioenergy is leading to increasing rates of LUC (Poeplau et al., 2011). We assume that *i)* in the differentiated approach, carbon dynamics in the soil follow an exponential decrease across time in line with Poeplau et al. (2011), and that *ii)* biomass-related emissions are instantaneous.[23] Since this paper is mainly addressed to European policy-makers, we use a 20-year time horizon for LUC emissions as assumed in the European RED. The discount rates we employ are constant[24] and range from 0 to 5% in the analysis, which is in line with the estimated values of the discount rate found in cost-benefit analyses of public projects and policies in Europe ((Florio, 2014), p.187). We consider three scenarios: a 0% discount rate as the baseline, a 3% discount rate as recommended by the European Commission (2014) in the EU funds framework[25] and a 4.5% discount rate as recommended by Quinet (2013) and France Stratégie (2017) for the evaluation of public investment projects in France. Finally, for the sake of clarity in this subsection, we consider carbon prices that grow at a constant rate[26] close to average growth rates that can be found in existing carbon price scenarios. We consider an initial price of 87€ in 2020 as recommended by Quinet (2019). The initial carbon price is kept constant across scenarios for the sake of comparability. Each scenario is characterised by a specific carbon price growth rate as follows:

- Scenario O: 0%, baseline scenario with constant carbon prices over time;
- Scenario A: 3%, close to the average growth rate of the carbon price in the Current and New Policy Scenarios in the World Energy Outlook (IEA, 2018);
- Scenario B: 4.5%, carbon price growth rate considered between 2040 and 2050 in the Quinet (2019) report. This is also the current discount rate employed in French public project assessment, which allows us to discuss the Hotelling rule;
- Scenario C: 6%, close to the average growth rate of the carbon price in the Sustainable Development Scenario in the World Energy Outlook (IEA, 2018) and in OECD (2018b).

Because the initial price is assumed to be the same across all scenarios, environmental objectives are considered increasingly constraining from Scenario O to Scenario C. In addition to Scenarios O, A, B and C, we consider the carbon price trajectory of the Quinet (2019) report, henceforth shadow price of carbon (SPC) scenario, the carbon price growth rate of which is not constant over time (see Table 1). For the sake of comparison between Scenario SPC and Scenarios O, A, B and C, the average annual growth rate of the carbon price in the SPC scenario is 9.1% between 2020 and 2040, the period over which we consider the biofuel project.

3.2.2. Data

The computation of LUC emissions relies on the formal definitions of the uniform and differentiated approaches as described in Appendix

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[22] Grassland ploughing has increased in the years 2000 in particular because of an increase in agricultural prices (Chakir and Vermont, 2013). Despite the regulations prohibiting the conversion of high-carbon land types, grasslands and some forestlands continue to be ploughed and cleared due to the considerable incentive to develop energy crops (ibid). Unfortunately, in France, the available data on agricultural areas does not allow us to distinguish the effect of energy-related land conversions from that of food-related ones (ibid). In their recommendations, Chakir and Vermont (2013) mention that the conversion of grassland to energy crops remains the most important source of LUC emissions related to the development of biofuels in France.

[23] Nonetheless, the rate of decay of the initial biomass depends on how it is managed afterwards, e.g. whether it is left to decompose or is burned, buried or converted into long-lived products such as furniture (Delucchi, 2011). This is taken into account through the variables ω_s and ω_v described in Appendix A.2.

[24] We discuss this assumption in Section 5.

[25] It is indeed possible that biofuel projects are funded by different member states in the European Union.

[26] This assumption might presently be restrictive since most existing carbon price scenarios, which we explore in Section 4 (see Table 1), do not entail a constant carbon price growth rate. One explanation for the absence of constant rates in these scenarios lies in the fact that climate objectives are becoming increasingly stringent, requiring a smoothing of carbon price trajectories from relatively low current prices until they reach a point where they align with environmental targets (Quinet, 2019, p.122).

A.2. To determine carbon stock changes in soil and vegetation, we rely on the guidelines provided by the European Commission (2010), which are based on IPCC (2006). Such a calculation requires knowledge about climatic region, soil type, agricultural management, agricultural practices (input level) and crop yields. The assumptions on these factors for our case study are described in Appendix A.3. Regarding the share of carbon that is converted into CO_2 emissions, we assume that 30% of the carbon stock in soil is converted into CO_2 (as in Anderson-Teixeira et al., 2009). This figure falls in the range given by the Winrock database (see Table 1 in Broch et al. (2013)) and is very close to the assumption of 25% made by Tyner et al. (2010). We assume that the reverse conversion is symmetric. Regarding the carbon stored in vegetation, we hypothesise that 90% is converted into emissions - a figure in line with the CARB policy in the United States.[27] An overview of the data used in the study, including sources, is provided in Appendix A.3.

3.2.3. Computation tool

We develop a Python program[28] to generate the uniform and differentiated time distributions and calculate the NPV of the GHG emissions of bioethanol projects under the two time profiles. Once LUC emissions due to soil and biomass carbon stock changes as well as their dynamics over time are determined,[29] carbon releases are converted into CO_2 emissions according to Appendix A.2, and finally priced using one of the scenarios listed above. Regarding price scenarios, an algorithm extrapolates prices in an exponential way between two one-time carbon prices, which allows us to generate a complete trajectory of carbon prices over the time horizon considered, since only sparse carbon prices are provided in most scenarios, including the World Energy Outlook's (IEA, 2018, p.604). The program essentially returns all the environmental NPV types necessary for the analysis, i.e. types related to LUC emissions (under each type of time distribution), non-LUC emissions and total emissions from biofuel production (i.e. LUC + non-LUC).

3.2.4. Results

All results assume a conversion of grassland to cropland (wheat). Note that environmental NPVs are always negative throughout the results since we focus on a land conversion that generates emissions and thereby costs to society. Because there are no scale effects on emissions due to LUC from the production of one unit of bioethanol, for the sake of simplicity, we consider that 1 tonne of bioethanol is produced each year for 20 years.[30]

➤ *Discounting effect*

Fig. 2 illustrates the discounting effect for grassland converted to cropland. Carbon prices are constant over time and equal to 87€/tonne of CO_2.

When no discounting is applied (0%), the NPVs under the uniform and the differentiated approach are equal since points in time are affected in the same manner. When a 4.5% discount rate is applied, the uniform approach raises the NPV (or equivalently, drops the cost) of emissions due to LUC from −78.44€ to −53.31€ per tonne of bioethanol. By contrast, the differentiated approach does not change NPVs much under different discount rates because emissions are mostly upfront and therefore do not suffer much from the discounting process. The higher the discount rate, the larger the misestimation of the LUC-related NPV induced by the uniform time distribution, ranging from 23.15% for a 3% discount rate to 31.73% for a 4.5% discount rate.

➤ *Carbon price effect*

Fig. 3 illustrates the carbon price effect in the case of a conversion of grassland to cropland. Carbon prices are now increasing according to the different scenarios defined above (O, A, B, C) and the discount rate is zero. We also consider the shadow price of carbon (SPC) determined in the Quinet (2019) report since it is the reference for carbon values over time in France.

Fig. 3 shows that the NPV of emissions due to LUC is underestimated under the uniform approach (drops from −78.44€ to −223.02€). The higher the carbon price growth rate (from Scenario O to Scenario SPC), the larger the bias induced by the uniform approach (downward bias ranging from 33.93% under Scenario A to 180.97% under Scenario SPC).

➤ *Combined effect*

When combining a positive discount rate (fixed to 4.5% in line with evaluations of public investment projects in France) with an (average) carbon price growth rate ranging from 0% (Scenario O) to 9.1% (Scenario SPC), the direction of the bias depends on whether the carbon price growth rate grows faster or slower than the discount rate (see Fig. 4).

In Scenario B, the Hotelling rule applies, which cancels the bias induced by the uniform approach. This is what should happen (theoretically after 2040) according to the Quinet (2019) report once carbon values have been revalorised according to the 1.5 °C limit on global warming. In Scenarios O and A, the discount rate is greater than the carbon price growth rate, hence the overestimation engendered by the uniform time distribution of 23.15% and 12.40%, respectively. In Scenarios C and SPC, the carbon price grows faster than the discount rate, which makes the uniform approach distort the cost of emissions upwards. The LUC-related NPVs are underestimated by 14.71% and 69.79% respectively.

It is worth highlighting here that these results only apply for direct LUC. But the (physical) mechanism of land conversion is the same whether LUC is direct or indirect, which means that the present (already substantial) bias is underestimated compared with an analysis also incorporating indirect LUC. We discuss this further in Section 5.

4. Proposal of two simple tools for decision-makers

Given the NPV misestimation that the uniform approach induces, we provide two simple tools to help decide whether to implement a biofuel project, namely the compensatory rate (Section 4.1) and the carbon profitability (CP) payback period (Section 4.2). Our tools exclusively rely on the environmental, i.e. non-market-related, part of CBA for several reasons. First, because CBA is monetary per se and thus aggregates monetised environmental flows with market flows, the economic NPV, i.e. market-related,[31] would just be translated by the environmental NPV downwards (upwards) in the case of net emissions (sequestrations) related to the project. Therefore, the environmental NPV, calculated by the Python program available online, can simply be added to the economic NPV. Second, the economic part of CBA relies on multiple (private) determinants such as land prices, competitive advantage and political context. By contrast, the environmental part of

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[27] Tyner et al. (2010) and Searchinger et al. (2008) assume that 75% and 100% is converted into emissions, respectively.

[28] Namely PyLUCCBA. The program (complete tool coded in Python language) is publicly available on GitHub, *https://github.com/lfaucheux/PyLUCCBA* and on Pypi, https://pypi.org/project/PyLUCCBA/. The program is also described in the supplementary material linked to this paper.

[29] Referring to Appendix A.2 regarding the differentiated approach (Definition 2), the program determines the coefficient a of the carbon response function provided by Poeplau et al. (2011), while taking into account the associated time horizon (for soil or vegetation).

[30] Of course, this trajectory can be changed in the Python program in order to obtain NPVs associated with a specific project.

[31] I.e., not related to social considerations.

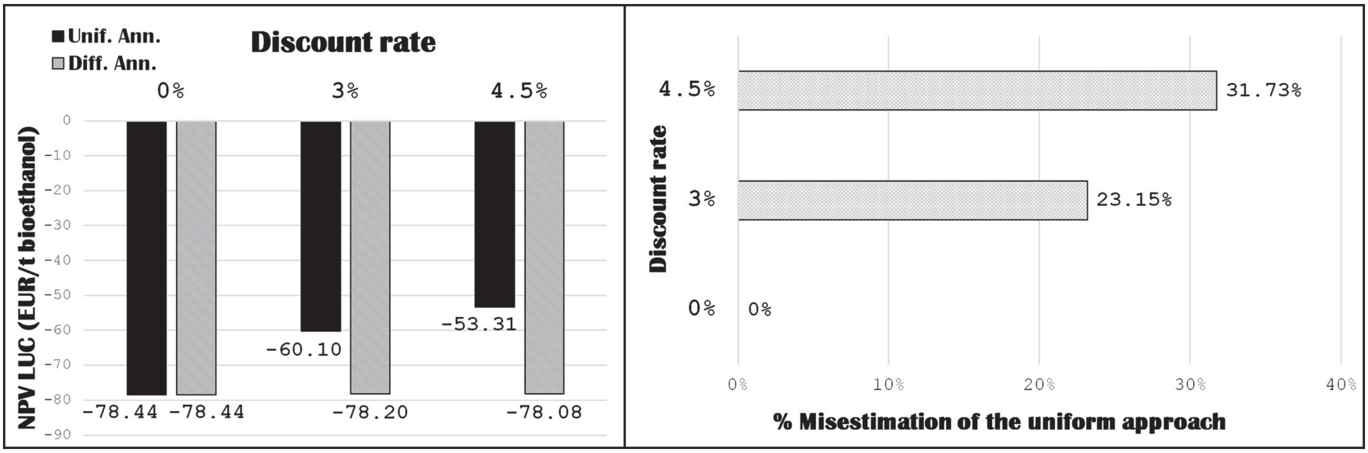

Fig. 2. Net Present Value of LUC emissions (left) and relative upward bias induced by the uniform approach (right) for different discount rate values. For grassland conversion.

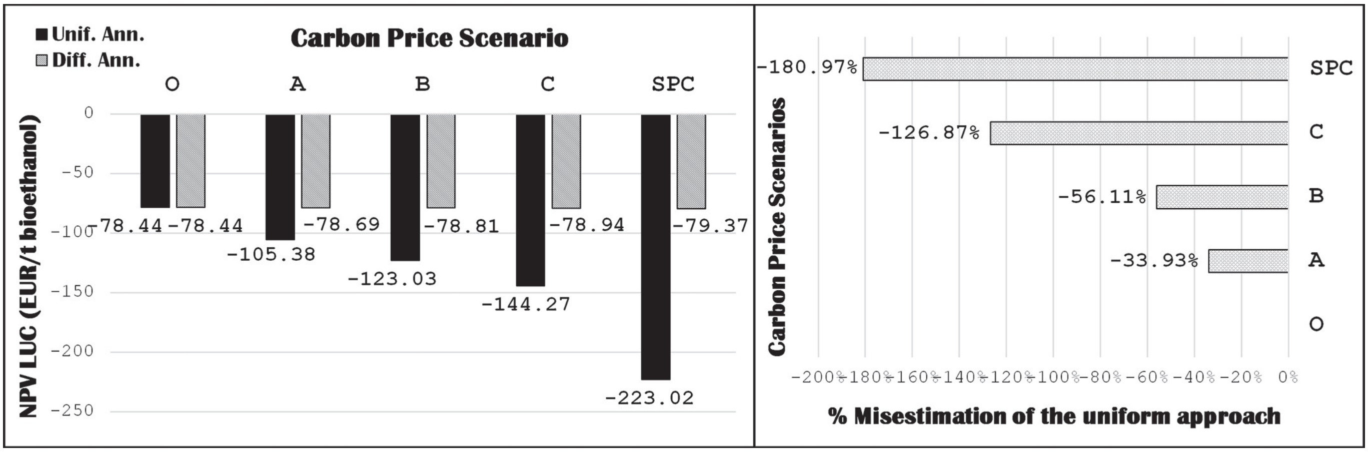

Fig. 3. Net Present Value of LUC emissions (left) and relative downward bias induced by the uniform approach (right) for different carbon price scenarios. For grassland conversion.

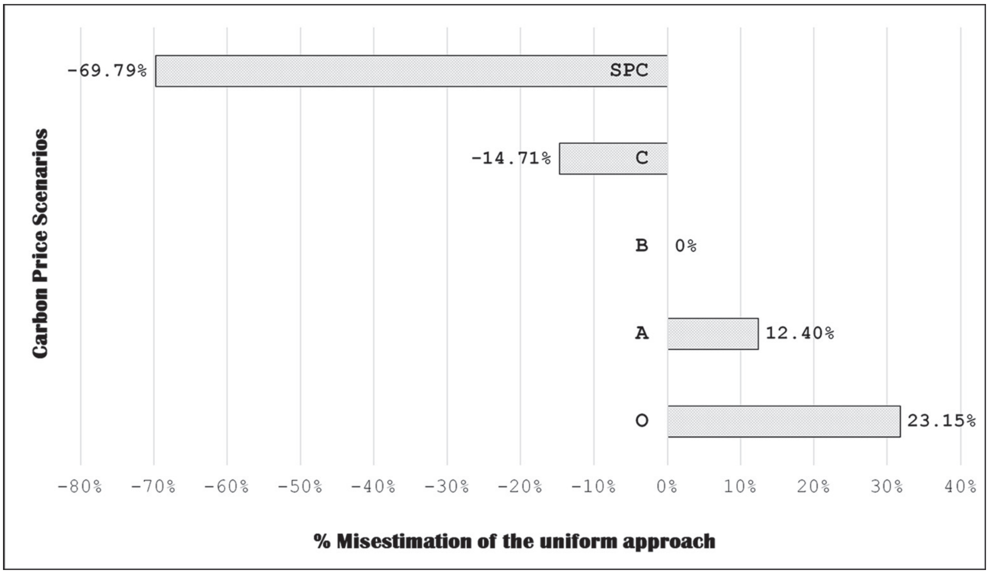

Fig. 4. Relative bias induced by the uniform approach (4.5% discount rate and different carbon price scenarios). For grassland conversion.

CBA is independent of the project holder's specificities and relies on isotropic determinants such as the conversion rate of carbon fluxes into carbon emissions and standard carbon price trajectories.[32] The particularities of the environmental part of CBA are all incorporated in the Python program developed for the purpose of this study and, more generally, decision-making. All specificities can be changed or enriched[33] according to the project at hand, e.g. crop type and its consequences on carbon stock changes and emissions from cultivation and production processes. Third, the capacity of the atmosphere to handle GHG emissions is limited, which makes the consideration of the environmental part of CBA interesting. The traditional use of payback periods of a project in economic calculation is informative, but we argue that it could be complemented with carbon-specific payback periods as presented in Section 4.2, if one wishes to emphasise environmental concerns in the CBA context.

4.1. Compensatory rate

We define the compensatory rate as the discount rate value that cancels the bias induced by the uniform approach given a carbon price path. Put differently, it is the rate that equalises NPVs under the uniform and differentiated approaches. While such a concept may seem trivial if we consider that carbon prices grow at a constant rate (as assumed in our theoretical model), the compensatory rate is of particular interest when using existing carbon price paths (e.g. OECD, 2018b; Quinet, 2019) in which carbon prices do not grow at a constant rate.[34] The compensatory rate depends on both the carbon price path and the time distribution of emissions (to which carbon prices apply).

We consider different carbon value trajectories, including the SPC scenario (Quinet, 2019), which is the reference for carbon values in France and complies with the latest 2018 IPCC report range of values, the OECD scenario reported in the questionnaire addressed to OECD countries on the current practices of CBA for public investment projects (OECD, 2018b) and the Current Policy Scenario (CPS), New Policy Scenario (NPS) and Sustainable Development Scenario (SDS) i.e., the three trajectories from the World Energy Outlook (IEA, 2018). These five scenarios, which carbon price growth rate is not constant over time, are likely to be used in project assessment in France and Europe. Those are presented in Table 1.

As can be observed in Table 1, the Quinet (2019) report has the most constraining carbon price trajectory compared with the other scenarios.[35]

Table 1
Carbon price scenarios (in € 2018).

Carbon price scenarios	2016	2020	2025	2030	2040	2050
Quinet (2019)		87		250	500	775
OECD (2018b)	62.7	78.8		139.1		335.6
Current Pol. Sc. IEA (2018)			25.4		43.8	
New Pol. Sc. IEA (2018)			28.8		49.6	
Sustainable Dev. Sc. IEA (2018)			72.6		161.4	

The compensatory rate[36] serves as a benchmark for the discount rate chosen in a project evaluation. If the compensatory rate is lower (higher) than the discount rate chosen in CBA, it informs decision-makers that the value of the project will be overestimated (underestimated). Therefore, this tool provides information about the direction of the estimation bias due to the use of the uniform time distribution given a specific carbon price trajectory. Fig. 5 provides a numerical illustration of the compensatory rate applied to the carbon price trajectories described in Table 1 in the context of bioethanol production in France (related to the conversion of grassland to cropland). The more constraining the scenario, the higher the compensatory rate.

Let us look at the current project evaluation practice in France, i.e. utilisation of the uniform approach with a 4.5% discount rate to discount future emissions. Using the SPC, OECD and SDS scenarios leads to an overestimation of emissions costs (or equivalently an underestimation of the NPV of LUC-related emissions), while using the CPS or the NPS scenario results in an underestimation of emissions costs. The higher the gap between the compensatory discount rate and the discount rate used in CBA, the larger the misestimation.

4.2. Carbon profitability (CP) payback period

The second tool to help decide whether to implement a biofuel project relies on the whole environmental part of CBA, i.e. on including LUC and non-LUC emissions. Non-LUC emissions encompass emissions from the production, transport and distribution of biofuels and the cultivation of energy crops. As in Section 3, we consider land conversion from grassland to wheat fields. Bioethanol projects are compared with fossil fuel production projects based on equivalent amounts of energy produced. In this context, GHG savings are allowed because aside from LUC emissions, the amount of GHGs emitted from the production and consumption of fossil fuels is greater than the energy-equivalent GHG amount from bioethanol production and consumption.

We introduce the concept of monetised carbon investment, which is illustrated in Fig. 6 (bottom chart) for the SPC scenario.[37] This concept only holds for the differentiated approach. Under the uniform approach, emissions are spread out over 20 years, which does not make clear the initial carbon investment that, in contrast, the differentiated approach involves. Land conversion simulates a (shadow) carbon investment since upfront emissions constitute a social cost incurred at $t = 0$ that is refunded through future GHG savings (hence relative carbon benefits). These future GHG savings are expected to counterbalance the initial cost at the so-called CP payback period. The monetised carbon investment could also be considered as a borrowed (monetised) amount of carbon from the atmosphere that is returned in the future. It is worth mentioning that it differs from the widespread ‘carbon debt’ concept by its being monetary and not physical (i.e. emissions quantities are priced here). In Fig. 6, we plot environmental

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[32] Often specific to a whole region or country.

[33] Indeed, the tool is publicly available and developed with an intention to promote future collaborative work on the tool itself or the data chosen to conduct new numerical exercises.

[34] If carbon prices grow at a constant rate, equalising the NPVs of the uniform and the differentiated approach amounts to discounting emission flows with the rate equal to the constant(or equivalently average) carbon price growth rate. This means that the compensatory rates of Scenarios O, A, B and C are 0%, 3%, 4.5% and 6% respectively. If carbon prices do not grow at a constant rate, discounting emission flows with a rate equal to the average carbon price growth rate does not equalise the two NPVs. This is because the average annual growth rate of carbon prices only considers the carbon prices in the first and last years of the project, thereby neglecting the effective trajectory of prices between these two years. Therefore, the compensatory rate should not be confounded with the average growth rate of a carbon price trajectory.

[35] The OECD survey related to the current practice of CBA in the transport and energy sectors was addressed to OECD countries in 2016. This was before the conclusions of the IPCC report on the limitation of global warming to 1.5 °C, which updated reference carbon values ((Rogelj et al., 2018) IPCC report). These conclusions are taken into account in the shadow price of carbon of the (Quinet, 2019) report. We can expect the carbon values in current practices of CBA to be updated in the near future in line with the Quinet report and therefore the 2018 IPCC report.

[36] Calculated by the Python program described in the supplementary material and available on GitHub.

[37] Note that in the differentiated approach, the initial kink on every curve is due to the one-year delay of biofuel production. LUC occurs at $t = 0$ and the process of production that allows for ‘GHG refunding’ starts at $t = 1$.

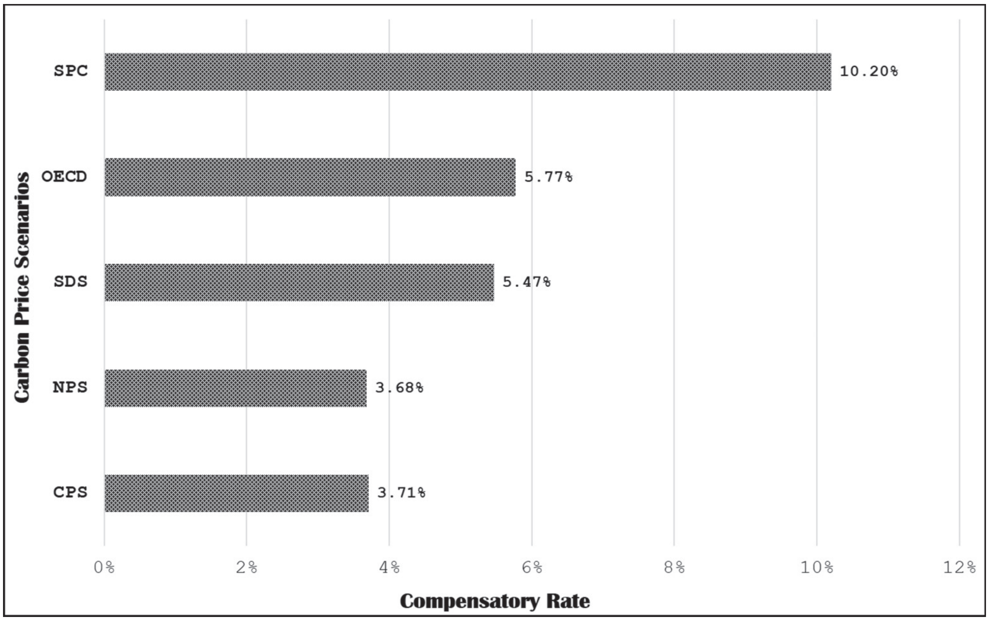

Fig. 5. Compensatory rate across different carbon price scenarios, conversion of grassland to cropland.

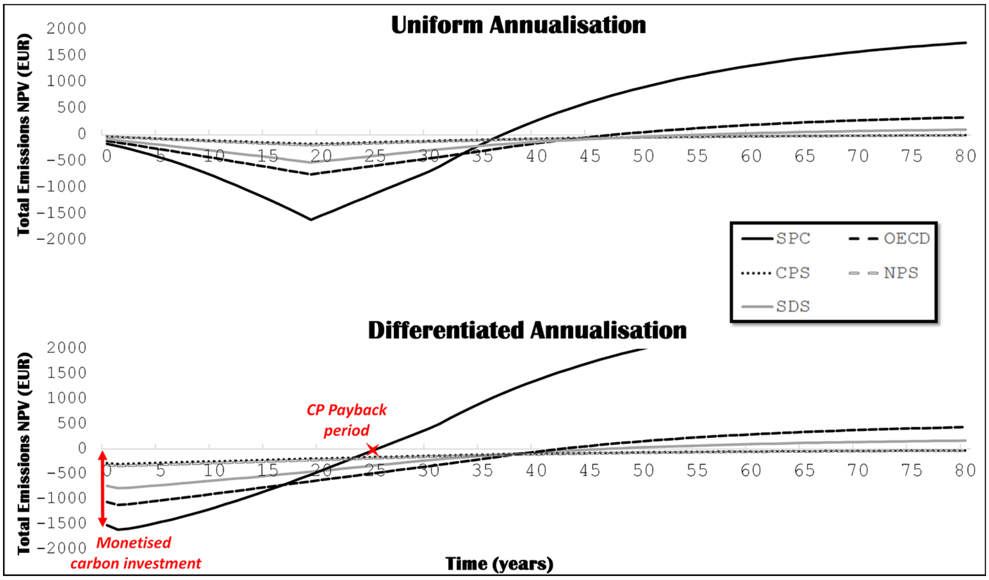

Fig. 6. Carbon profitability payback periods across different carbon price scenarios under the uniform (top chart) and the differentiated (bottom chart) time distribution.

Table 2
Carbon profitability payback period across carbon price scenarios and time distributions.

	Uniform	Differentiated
CPS	102	> 200
NPS	106	> 200
SDS	55	47
OECD	48	43
SPC	37	26

NPVs under both the uniform and the differentiated approach for the common price scenarios described in Table 1.

The CP payback period changes across scenarios and across time distributions as reported in Table 2.[38] Overall, all payback periods are relatively high (higher than the time horizon of the project). The payback periods computed under the uniform approach for the SPC, OECD and SDS scenarios are greater than those under the differentiated approach. By contrast, the payback periods computed under the uniform approach for the CPS and NPS scenarios are smaller than those under the differentiated approach.

The problem with using the uniform approach is that an LUC-related project may pass the CBA test under the differentiated approach but not under the uniform approach or vice-versa. If decision-makers use a benchmark CP payback period, which should be pre-established by

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[38] CP payback periods are also calculated by the Python program, which is described in the supplementary material and available on GitHub.

policy-makers,[39] this benchmark could be compared to the CP payback period of projects. E.g. in the SPC scenario, if the benchmark were fixed to 30 years, the project would not pass under the uniform approach while in reality (i.e. under the differentiated approach), emissions do comply with such a requirement, thereby penalising projects that would actually be considered as beneficial to the environment according to predetermined benchmark. By contrast, with the NPS scenario, where the carbon price grows slower than the discount rate, the uniform approach may end up lending support to projects that are actually harmful to the environment.[40] Therefore, the CP payback period addresses the issue of decision error when mainly or partly based on CBA. The uniform approach may either be at odds with the primary objective of cutting emissions by not rejecting environmentally harmful projects or lead to the disapproval of projects that actually comply with the requirements (e.g. the benchmark payback period).

In addition to the consideration of complete cost-benefit analyses that enable the calculation of general payback periods of investment projects, the environmental part alone *should* inform decision-makers about environment-specific payback periods as a complementary tool. This is all the more relevant in a policy context that needs to comply with more stringent environmental objectives as required by the 2018 IPCC report.

A limitation of this tool may be the absence of consideration of potential scale effects in biofuel production. Indeed, the carbon profitability payback period also involves non-LUC emissions from the production process, and thus, it is subject to economies of scale (which is not the case for LUC emissions). Intuitively, taking these economies of scale into account would shorten the estimated payback periods for both time distributions since economies of scale lead to higher energy efficiency in biofuel production and thus faster net GHG savings across the whole project time horizon. Nevertheless, nothing would change the aforementioned conclusions regarding the comparison between the uniform and the differentiated approach.

5. Discussion

In this section, we discuss our assumptions and extend the implications of our results to further issues such as indirect LUC, the accounting for sequestrations often linked to second-generation biofuels and the consideration of LUC impacts in carbon markets.

5.1. The CBA framework

Cost-benefit analysis is a decision-support tool that is widely used in project evaluation (OECD, 2018b). Its popularity can partly be attributed to its convenience and simplicity in aggregating market flows with priced (non-market) CO_2 flows, resulting in a synthetic assessment indicator, i.e. the NPV of the project. Nonetheless, we do not argue that it is the only approach that should be used in project assessments. Instead, we emphasise that such a widespread tool, whose influence on final decisions varies from moderate to large (OECD, 2018b), should be used with caution when environmental impacts are characterised by a peculiar time profile like LUC. Cost-benefit analysis should not be considered a unique answer to project assessment, especially when other environmental impacts (on e.g. biodiversity or water, the monetary valuation of which may not exist or may not be as robust as carbon values) are affected by the project ((OECD, 2018b), Figure 16.9). This economic tool should be complemented with other approaches such as multicriteria analysis that can account for dimensions beyond e.g., economic efficiency (OECD, 2018a). Overall, “the role of CBA remains one of explaining how a decision should look if the economic approach is adopted” (OECD, 2018c). This paper aims at promoting tools that, although economic, try to be consistent with biophysical reality. Still, greater consideration should be given to the interdisciplinarity of approaches because it allows for a broader picture of the consequences of the implementation of a project.

5.2. Discounting and time horizon assumptions

Exponential discounting was assumed, in line with the practice of project assessment guidelines suggested in European policies, i.e. within a 20-year time horizon. While our objective was to raise the issue of not considering correct land use change dynamics in *current practices* of socio-economic analysis, both assumptions on the discount rate and the time horizon are worth discussing. First, such a short time horizon is generally chosen to fit the expected duration of biofuel production. It has the advantage of emphasising the importance of large upfront emissions due to land conversion. Yet, it does not account for *(i)* the persistence of GHGs in the atmosphere for long periods, *(ii)* the future of energy cropland (e.g., land reversion) and *(iii)* intergenerational issues. In a way, using a short time horizon is a ‘conservative’ approach since longer time horizons come with growing uncertainty (Broch and Hoekman, 2012). Besides, there is a large debate around the value and trajectory of the discount rate over time. While some economists are in favour of discounting environmental values, others are more reluctant to the idea. Still, economists agree on the need to reconcile discounting with sustainability and intergenerational equity (Martínez-Paz et al., 2016). Discounting relies on two main arguments: *(i)* individuals have a pure preference for the present and *(ii)* future generations are expected to be richer than today, increasing consumption inequalities over time (Gollier, 2002). No or low discounting gives more weight to the well-being of future generations. Within CBA, the objective is to apply the Hotelling rule to prevent discounting from overwhelming the value of emissions in the future. However, the rule is currently hardly applicable because of the gap between current carbon prices and those that should reflect objectives of global warming limits (Quinet, 2019). Despite the lack of consensus on the suitable value for discount rates, the use of declining discount rates, as introduced in France by the Lebègue (2005) report for public investment projects, has become more common under longer time horizons (Guesnerie, 2017). This allows one to put less weight on the longer term, which is characterised by uncertainty surrounding both economic growth and long-term environmental impacts (see e.g., Arrow et al., 2013, 2014). In France, the declining profile of discount rates is effective only 30 years after the project starts, which we did not explore in our numerical illustration as it considers 20 years as the time period over which ethanol production and LUC impacts should be examined (Delucchi, 2011; European Commission, 2010; IPCC, 2006). Nonetheless, when using the CP payback period, a benchmark in excess of 30 years would justify the use of declining discount rates in our calculations.

5.3. Extension to indirect land use change

Because of the uncertainty surrounding the identification and quantification of indirect LUC (Di Lucia et al., 2012), we only dealt with direct LUC. However, the philosophy behind the model can apply to any phenomenon that entails the same carbon dynamics, thereby including indirect LUC.[41] It is worth emphasising that the magnitude of the bias can be expected to increase with the accounting of indirect LUC, which is currently a central issue in European policies (European Commission, 2015b, 2018b). Besides uncertainty, indirect LUC differs from direct

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[39] For example, the benchmark could require that the payback period is lower than the time horizon of the project.

[40] In this case, the benchmark payback period would be violated under the differentiated but not the uniform approach.

[41] Provided the information on carbon stock changes related to indirect LUC, the Python program we developed in this paper can accommodate such impacts.

LUC in terms of the stage of a project at which it arises. Indeed, Zilberman et al. (2013) point out that indirect LUC occurs with significant time lags. Empirical evidence suggests that the materialisation of indirect LUC takes 10–15 years after land is converted to energy crop fields (Andrade De Sá et al., 2013). This implies that, with a 20-year time horizon, a potentially large part of indirect LUC emissions related to a project would be truncated in CBA. Indeed, like direct LUC emissions, indirect LUC emissions should be considered over a 20-year time period as suggested by IPCC reports. The application of the uniform approach would strongly affect the accounting of indirect LUC emissions because all emissions above the time horizon (i.e. constant emissions over 5–10 years) would not be considered. If instead, the differentiated approach applies to indirect LUC emissions, most emissions would be accounted for within the period over which the project is considered. Therefore, the use of the uniform approach for indirect LUC emissions would enhance the misestimation of the NPV for two reasons: *(i)* the larger truncation of emissions under the uniform approach than under the differentiated approach and, *(ii)* the fact that emissions under the uniform approach undergo the discounting and carbon price effects more than under the differentiated approach. If one wishes to consider the entirety of carbon-related impacts of a biofuel project, an adaptation of the time period over which biofuel production projects are assessed is necessary when indirect LUC emissions are considered in CBA.

5.4. Second-generation biofuels and carbon sequestrations

While the focus of our paper was LUC emissions, our framework could also apply to LUC sequestrations.[42] Second-generation biofuel projects are particularly promising for carbon sequestration (see e.g. Anderson-Teixeira et al., 2009; Nakajima et al., 2018) conditional on energy crops not replacing lands with higher carbon content (Don et al., 2012). There are a growing number of second-generation biofuel projects in France, e.g. Futurol and BioTfuel. However, the dynamics of LUC sequestrations are less clear than those of LUC emissions. The results of the meta-analysis by Qin et al. (2016) and the study by Poeplau and Don (2014) suggest that sequestrations are not constant over time and might not even be monotonic, thereby questioning again the uniform time distribution assumption currently adopted in European policy.[43] Provided the knowledge of the correct time distribution of sequestrations, only the compensatory rate would be useful to support decisions. Indeed, sequestrations constitute a benefit to society, which makes the use of CP payback periods irrelevant.

5.5. LUC dynamics and carbon markets

Reductions of emissions from LUC are part of the 2018 European Union climate legislation for the period 2021–2030 (European Commission, 2018b). Although LUC considerations are not covered by the European Emission Trading Scheme (ETS) (Hamrick and Gallant, 2017; ICAP, 2019),[44] the implications of the discrepancy between LUC temporal profiles under this widespread quantity-based instrument are worth discussing. Currently, the sectors covered by the EU ETS, e.g. energy, purchase permits in line with their effective annual needs. This would not be the case under the uniform approach that does not reflect the real dynamics of LUC emissions. If LUC emissions were capped, the consideration of the uniform approach would allow biofuel producers to smooth their need for emission permits over time. However, biofuel producers would also suffer from increasing prices over time. If instead the differentiated temporal profile were adopted, biofuel producers would not be able to smooth their need for carbon allowances over time. They would most likely need to purchase permits in the early phase of production, the upfront purchase potentially weighing heavily in their cost-benefit balance depending on carbon market prices.

6. Concluding remarks and policy recommendations

This paper built on the confrontation between scientific evidence and policy assumptions regarding the temporal profile of LUC emissions. We examined the consequences of using the uniform time distribution approach in project assessment when CBA is used. While we acknowledge that the sole use of CBA approach can be questioned (Norgaard, 1989), at least, when used to assess LUC impacts, it should be done properly. We found that distortion of NPVs occurs upwards (downwards) if the carbon price grows slower (faster) than the discount rate. While our results apply to all countries under European policy,[45] we illustrated them with the case of French bioethanol production. We estimated that using the uniform distribution leads to an overestimation of direct LUC emission costs by up to 70% for wheat-based ethanol in France. This result could lead to the non-implementation of such a project despite actual compliance with environmental requirements. We provided two simple tools to help decision when faced with such an issue. The compensatory rate indicates the direction of the misestimation given the specificities of the project and parameters of the CBA. The carbon profitability payback period suggests a price-based carbon-specific payback period for the project that could be compared with a benchmark predetermined by policy-makers.

The objective of this paper was to raise the *current* accounting for LUC dynamics in European policy and the problem it might cause in project assessment when CBA is used. Economic processes, reflected in CBA, treat different points in time differently through the use of discount rates and increasing carbon prices whereas policy assumptions, often based on life-cycle assessment results, uniformly amortise LUC emissions over time. Our first-best recommendation, specifically addressed to policy-makers, is to correct for this disconnection in policy assumptions by relying more on academic research on the dynamics of LUC. This would avoid misleading NPV results when CBA is used as a decision-support tool. If the available empirical evidence (e.g. Poeplau et al., 2011) is deemed insufficient, a reasonable alternative that is closer to the biophysical reality than the uniform approach would be to consider that the total emissions from biomass removal in connection with land conversion are felt immediately instead of spread evenly over time. It is worth mentioning that the US biofuel policy (RFS2) has gone a step forward (compared with the European Union) by disentangling the two carbon sinks (soil and biomass): biomass-related LUC emissions are fully accounted for at the time of land conversion while soil-related LUC emissions are uniformly distributed over time. A broader classification of the time distribution approaches used by policy-makers is provided in Appendix A.2. Still, since the recent Renewable Energy Directive reiterated the uniform time distribution assumption (European Commission, 2018a), we recommend the use of the two tools suggested in this paper in the context of project assessment to complement traditional CBA results.[46] The compensatory rate and the carbon profitability payback period are provided by the online Python program once a project of interest has been specified. The program

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[42] For example, a conversion of cropland to farming of *Miscanthus* harvested for ethanol production.

[43] The Python program, available online, can be used in the case of LUC sequestrations provided that carbon response functions are adapted to the land conversion under study in the code. Indeed, the current carbon response function relies on an exponential decline of SOC based on Poeplau et al. (2011), which may not apply to sequestrations. The program was conceived and organised with the intention of making any assumption change easy.

[44] A few countries or regions such as New Zealand do account for agriculture and forestry in their domestic ETS (Hamrick and Gallant, 2017; ICAP, 2019).

[45] Most of which use CBA for project assessment (OECD, 2018b).

[46] This recommendation is primarily addressed to public decision-makers but also private decision-makers who need to comply with increasingly constraining environmental objectives.

allows (public or private) decision-makers to obtain the environmental part of their project's NPV, which can easily be added to the economic part. Both the compensatory rate and the carbon profitability payback period are adapted to the current policy situation and are therefore necessary while waiting for the transition towards more consideration of LUC dynamics in policy.

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Appendix A. Appendix

A.1. Proof of Proposition 3

$$\Delta NPV = NPV_u - NPV_d \tag{A.1}$$

$$= -\left(p_0\frac{z_0+z_1}{2} + p_1\frac{z_0+z_1}{2(1+r)}\right) - \left(-p_0 z_0 - p_1\frac{z_1}{1+r}\right) \tag{A.2}$$

$$= -\frac{p_0(z_0+z_1)(1+r) - p_0(1+g)(z_0+z_1) + 2p_0 z_0(1+r) + 2p_0(1+g)z_1}{2(1+r)} \tag{A.3}$$

$$= -\frac{p_0}{2(1+r)}(z_0(g-r) + z_1(r-g)) \tag{A.4}$$

$$\Delta NPV = \frac{p_0}{2(1+r)}(z_0 - z_1)(r-g) \tag{A.5}$$

Since by assumption $z_0 > z_1$, the sign of ΔNPV only depends on the sign of *r*-*g*.

A.2. LUC emissions time distribution: formal description

The following formal definitions of the uniform and differentiated approaches are implemented in the Python program to generate the numerical results provided in Sections 2 and 4.

Let us denote by *SOC* and *VGC* the carbon stocks in soil and vegetation (biomass), respectively, expressed in tonnes of carbon per hectare. Then, $\Delta SOC = SOC_F - SOC_I$ and $\Delta VGC = VGC_F - VGC_I$ are the carbon stock differences between land conversion and equilibrium achievement where *I* and *F* refer to initial (before conversion) and final (after conversion) lands, respectively. z_t is expressed in tonnes of CO_2 per unit, e.g. hectare or tonne of ethanol, per year. z_t is decomposed into z_t^s and z_t^v the *annual* LUC emission flow from soil and vegetation, respectively. z_t^s and z_t^v are respectively spread out over the time horizons T^s and T^v. ω_s and ω_v are introduced as the respective shares of soil and vegetation carbon that are converted into CO_2 emissions.[47] *A* is a constant that includes at least the coefficient of conversion of carbon into CO_2.[48]

Definition 1. (**uniform annualisation**) *LUC emission flows are uniformly annualised* $T^v \le T^s$ *and emissions due to soil and vegetation carbon releases are constant over time i.e.* $z_t^s = z_{t+1}{}^s \,\forall\, t \le T^s$ *and* $z_t^v = z_{t+1}{}^v \,\forall\, t \le T^v$. *Then, the total annualised LUC emission is*

$$\forall\ t = \{0, 1, \ldots, T^s\},\ z_t = z_t^s + z_t^v = A\left[\omega_s\frac{\Delta SOC}{T^s} + \omega_v\frac{\Delta VGC}{T^v}\right]$$

with $\mathrm{z_t^v} = 0\ \forall\ \mathrm{t} \ge \mathrm{T^v}$.

Definition 2. (**differentiated annualisation**) *LUC emission flows are "differentially" annualised when* $T^v \le T^s$, $z_t^s \ne z_{t+1}{}^s \,\forall\, t \le T^s$ *and* $z_t^v \ne z_{t+1}{}^v \,\forall\, t \le T^v$. *Then, the total annualised LUC emission is*

$$\forall\ t = \{0, 1, \ldots, T^s\},\ z_t = z_t^s + z_t^v = A(\omega_s\, \Delta\mathrm{SOC}{\cdot}f_s(t) + \omega_v\, \Delta\mathrm{VGC}{\cdot}f_v(t))$$

with $\mathrm{z_t^v} = 0\ \forall\ \mathrm{t} \ge \mathrm{T^v}$.

f^s and f^v are continuous and monotonic functions of time that underlie the carbon response of soil and vegetation, respectively, to land conversion.

For a grassland or a forestland converted into a cropland, SOC decreases exponentially according to the meta-analysis of Poeplau et al. (2011).[49]

Definition 3. (**weak and strong definitions of LUC time distributions**) *The uniform and differentiated annualisations are characterised by the exclusion and inclusion of a carbon stock dynamics. The distinction between weak and strong definitions of LUC time distributions relies on whether* $T^v < T^s$ *or* $T^v = T^s$ *as described in* Table 3.

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[47] Carbon losses may be deferred when carbon vegetation is stored in wood products such as furniture or buildings (Marshall, 2009; Tyner et al., 2010).

[48] Typically, $A = \frac{44}{12}$ (IPCC, 2006). For biofuel production, $A = \frac{44}{12k}$ where the constant *k* refers to the biofuel yield in tonnes of biofuel per hectare.

[49] Such that $f^s(t) = e^{-\frac{t-1}{a}} - e^{-\frac{t}{a}}$ where *a* is a constant. (Poeplau et al., 2011) estimate stock dynamics such that $\forall t$, $SOC_t = \Delta SOC\left(1 - \exp\left(-\frac{t}{a}\right)\right)$. My focus lies on flows, hence the flow from the soil at time *t* is $z_t^s = SOC_t - SOC_{t-1}$. Note that regarding vegetation carbon stocks, if $T^v = 1$ e.g. clearing a forest, no dynamics of carbon are considered since only one flow occurs at $t = 0$.

Table 3
Weak and strong definitions of LUC time distributions.

		Time horizons	
		$T^v < T^s$	$T^v = T^s$
Carbon dynamics	No	Weak uniform	Strong uniform
	Yes	Strong differentiated	Weak differentiated

Definition 3 allows us to categorise energy policies according to the time distribution they consider for LUC emissions. The uniform annualisation definition is strong in the sense that it is the extreme case of uniformisation: emission flows (from both soil and vegetation) are equal over the same time period. This is a far cry from the real dynamics of LUC. By contrast, the differentiated annualisation definition is strong in the sense that soil- and vegetation-related LUC emissions are distinguished in both their time horizon and their dynamics. The strong differentiated annualisation is the closest definition to what is described in the scientific literature. The European RED is based on the strong uniform annualisation definition with the assumption that $T^v = T^s = 20$, and the U.S. RFS2 policy is based on the weak uniform approach with $T^v = 1$ and $T^s = 30$.

A.3. Data

Table 4
Data used for the bioethanol case study in France.

About	Choice/value	Reference
Region	France	–
Biofuel	Bioethanol	–
Biomass 1st generation	Wheat	(Chakir and Vermont, 2013)
Project starting year	2020	–
Discount rates	From 0% to 5%	(Florio, 2014; Quinet, 2013)
Project time horizon	20, $t = 0$ land conversion Period of production: 20 yrs from $t = 1$ to $t = 20$	(European Commission, 2009a; European Commission, 2015a; European Commission, 2018a)
Carbon price projections	WEO trajectories, OECD questionnaire, Shadow price of carbon in France	(IEA, 2018; OECD, 2018b; Quinet, 2019)
Crop yields	Wheat: 7.5 t DM/ha *Miscanthus*: 16.5 t DM/ha	Agreste IFP energies nouvelles
Process yields	Wheat: 0.28 t eth/t DM *Miscanthus*: 0.32 t eth/t DM	IFP energies nouvelles
Climatic region	$\frac{1}{3}$ warm temperate dry $\frac{2}{3}$ warm temperate moist	See map in (European Commission, 2010)
Soil type	High activity clay soil	(European Commission, 2010)
Land cover options	Cropland, *Miscanthus*, improved grassland, degraded grassland, forest	–
Agricultural management	Wheat: 60% full tillage & 40% no till *Miscanthus*: no till	Agreste
Agricultural practices	Wheat: 70% high input without manure 30% with manure *Miscanthus*: medium input	Agreste
Coefficient shares carbon to CO_2	Emi: $\omega_s = 30\%$ and $\omega_v = 90\%$ Seq: $\omega_s = 30\%$ and $\omega_v = 100\%$	See Section 2 of the paper
Non-LUC emissions	Wheat *Miscanthus*	Biograce (Hoefnagels et al., 2010)
Gasoline emissions	87.1 g CO_2/MJ	Joint Research Centre (JRC WTT report Appendix 2 version 4a, April 2014)

Appendix B. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolecon.2019.05.017.

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