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The land use change time-accounting failure

Marion Dupoux*

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 upwards (downwards) when carbon prices grow slower (faster) than the discount rate. We illustrate our results with French bioethanol production. 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.

Keywords: land use change dynamics, cost-benefit analysis, public assessment of projects, discounting, relative carbon prices, biofuel

JEL Classification: D61, H43, Q15, Q16, Q48, Q54

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21 1 Introduction

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

42 In this paper, we investigate the disconnection between scientific and policy considerations
43 of the temporal profile of LUC emissions. Indeed, European policies assume that LUC emis-
44 sions, irrespective of type of carbon sink, have a *uniform* (constant) time profile (European
45 Commission, 2009a; European Commission, 2015b; European Commission, 2018a). What
46 are the consequences of such an assumption on the assessment of biofuel-related investment
47 projects? Shedding light on this question and suggesting tools to support decision-making in
48 this context are the two main objectives of the present paper.

49 The *ex ante* assessment of projects relies on a variety of approaches, e.g. multicriteria anal-
50 ysis, cost-benefit analysis (CBA), risk assessment and public participation, that complement
51 each other to support the decision of whether or not a project should be implemented. In prac-
52 tice, CBA is a widely used tool in the assessment of public investment projects in the energy

¹This 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.

53 and transport sectors (OECD, 2018b).² It is reported that the influence of CBA on the decision
 54 of whether to implement a project is moderate to large (ibid). Discount rate and time path of
 55 carbon prices are the two key elements of CBA. Both affect emissions at different times differ-
 56 ently except when carbon prices grow exactly at the discount rate, i.e. when the Hotelling rule
 57 applies. This rule prevents the discounting effect from overwhelming the value of emissions
 58 over time and is widespread in climate change modelling (e.g. Dietz and Fankhauser, 2010)
 59 and the determination of shadow carbon values (e.g. Quinet, 2009; Quinet, 2019, in the long
 60 term). Nonetheless, in current practice, carbon prices usually deviate from this rule (Hoel,
 61 2009), at least temporarily. This is because they need to reflect increasingly stringent objec-
 62 tives to curb greenhouse gas (GHG) emissions (for example, the goal of limiting the increase
 63 in average global temperature by 2° C became a goal of limiting the increase to 1.5° C (Rogelj
 64 et al., 2018, IPCC report)). This requires a progressive alignment with the Hotelling rule from
 65 current, relatively low, carbon prices.

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

76 We illustrate our results with the case of French bioethanol production from wheat. Because
 77 of the complexity of the quantification of indirect LUC (see e.g. Di Lucia et al., 2012), we
 78 focus on direct LUC,³ which accounts for approximately half of LUC emissions associated
 79 with wheat-based ethanol (Fritsche et al., 2010). Under the assumptions used in France for
 80 project assessment, i.e. a 4.5% discount rate (Quinet, 2013; France Stratégie, 2017) and the
 81 shadow price of carbon estimated in Quinet (2019, p.32), we find that the LUC-related NPV
 82 of a bioethanol project that entails a conversion of grassland into cropland⁴ is underestimated
 83 by almost 70%. We explore more carbon price scenarios and find that the misestimation of the

²This 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.

⁴Such land conversion is common in France (Chakir and Vermont, 2013).

84 value of LUC emissions ranges from -70% to +23%.

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

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

116 2 Background

117 In 2009, the European Commission imposed a mandatory goal for member states to ensure a
118 10% minimum share of renewable energies (and particularly a 6% share of biofuels⁶) in trans-

⁵The tool is available on GitHub, <https://github.com/lfauchaux/PyLUCCBA>.

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

119 port petrol and diesel by 2020 (European Commission, 2009a, Renewable Energy Directive
120 (RED)). Although the sustainability criteria of biofuels mentioned that the whole life cycle
121 of biofuels must be considered when assessed (European Commission, 2009b), the study by
122 Searchinger et al. (2008) pointed out the LUC issue and the extent to which it might result in
123 a worse carbon balance for biomass-based fuels compared with that for fossil-based fuels. As
124 LUC became critical to the determination of the carbon balance of biofuels (Fargione et al.,
125 2008), it led policy-makers to amend the 2009 RED in order to include the indirect LUC im-
126 pacts that biofuel projects might cause (European Commission, 2015b). In this section, we
127 review the literature on the dynamics of LUC as estimated in scientific literature (subsection
128 2.1) and as assumed in European energy policies (subsection 2.2). We then raise the issue of
129 the discrepancy between these two ways of accounting for LUC dynamics when it comes to the
130 assessment of public investment projects (subsection 2.3).

131 2.1 LUC emissions temporal profile in academic research

132 Land conversion results in carbon stock changes. The carbon balance disturbance occurs in
133 biomass and soil, both of which constitute important carbon sinks.⁷ Depending on the carbon
134 concentration in both the initial and the final land, land conversion can either release carbon,
135 generating CO₂ emissions to the atmosphere, or store carbon, leading to CO₂ sequestrations
136 from the atmosphere. The present paper tackles the issue of emissions but extends the discus-
137 sion to sequestrations in Section 5. The dynamics of carbon losses are sink-specific. While
138 the change in biomass carbon stock is in most cases instantaneous (Delucchi, 2011), changes
139 in soil organic carbon (SOC) stock occur over the course of several⁸ years until the carbon
140 stock reaches a new equilibrium (Marshall, 2009; De Gorter and Tsur, 2010; Delucchi, 2011;
141 Poepflau et al., 2011; Don et al., 2012). Measuring SOC is a complex task (Anderson-Teixeira
142 et al., 2009). Nonetheless, there is a large literature on the dynamics of SOC changes due to
143 LUC. Some *assume* certain carbon response functions, such as linear (e.g., Anderson-Teixeira
144 et al., 2009) or exponential (e.g., Evrendilek et al., 2004; Delucchi, 2011) SOC stock losses
145 over time. Others investigate the carbon response function that best fits SOC stock changes
146 for different land conversions by means of meta-analyses (e.g. Poepflau et al., 2011; Fujisaki
147 et al., 2015, in the context of temperate and tropical regions, respectively). The carbon response
148 functions developed by Poepflau et al. (2011), based on empirical data, have often been consid-
149 ered a reference for temperate zones in later studies (e.g., Nyawira et al., 2016; Li et al., 2018;
150 Searchinger et al., 2018). In particular, the conversion of both grassland and forestland to crop-

⁷Soil 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).

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

151 land is characterised by an exponential decrease in SOC stocks.⁹ Overall, empirical evidence
 152 suggests that, when a land accumulates and maintains carbon stocks better than another land,
 153 the conversion of the former to the latter results in carbon losses that tend to decrease over time.

154 2.2 LUC emissions temporal profile in EU policy

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

- 167 • Uniform temporal profile: constant time profile as assumed in European energy policies
 168 and described in this subsection.
- 169 • Differentiated temporal profile:¹⁰ decreasing time profile as reported in the biophysics
 170 literature (see subsection 2.1).

171 These two temporal profiles are illustrated in Figure 1, where land conversion occurs at time
 172 $t = 0$. Note that the sum of emissions under both temporal profiles is the same over the time
 173 horizon. Only the dynamics over time varies. The next section sheds light on the issue that
 174 may arise from the discrepancy between the two temporal profiles when it comes to project
 175 evaluation.

176 2.3 Why the policy’s disconnection from science matters in project assess- 177 ment

178 Project assessment relies on a variety of complementary tools such as CBA, multicriteria anal-
 179 ysis and risk assessment (OECD, 2018a). Investments in the energy sector are often informed

⁹The 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).

¹⁰I.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.

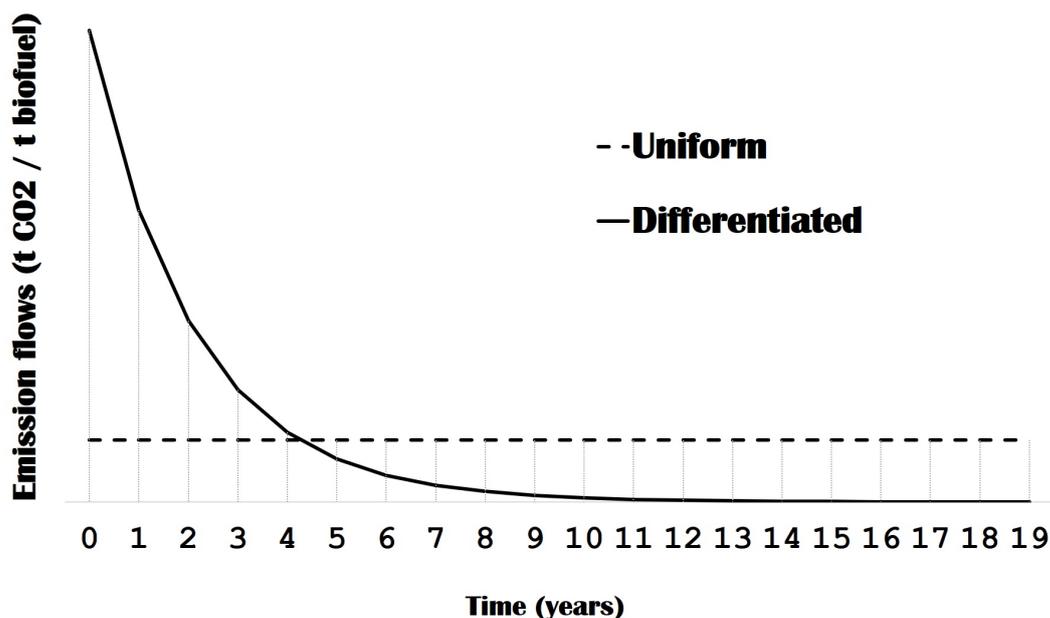


Figure 1: Temporal profiles of LUC emissions (uniform vs. differentiated).

180 by cost-benefit analyses that include GHG emissions (OECD, 2018b). In France, which is
 181 the country for our case study in subsection 3.2, socio-economic analysis is even mandatory
 182 (France Stratégie, 2017; Quinet, 2019, Box 10 p.139). In practice, final decisions are moder-
 183 ately or largely influenced by CBA results as reported in OECD (2018b, Figure 16.15). All
 184 these elements make CBA of biofuel projects worth regarding, especially when projects entail
 185 particular temporal dynamics like those of LUC emissions.

186 Cost-benefit analysis generally relies on *i*) pricing emissions at each point in time¹¹ and
 187 discounting future emissions costs over time.¹² Both carbon prices and the discount rate affect
 188 emissions differently over time.¹³ Only when carbon prices grow at the discount rate are emis-
 189 sions costs not affected by the time profile of emissions. This is known as the Hotelling rule,
 190 originally established for exhaustible resources.¹⁴ The Hotelling rule guarantees that carbon
 191 emissions do not suffer from discounting. Nonetheless, it is rarely the case that the discount
 192 rate employed in CBA of public investment projects is equal to the rate at which carbon prices

¹¹CO₂ 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).

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

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

¹⁴Applied 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 one tonne of CO₂ today or in a year, which in turn implies that the carbon price should increase at the discount rate.

193 grow over time (Hoel, 2009; Smith and Braathen, 2015; OECD, 2018b).¹⁵¹⁶ Indeed, while the
 194 Hotelling rule is considered a relevant rule in the long term, it is justified to temporarily get
 195 away from it to smooth the revalorisation of the climate action, and therefore the trajectory of
 196 carbon values over time (Quinet, 2019, p.123). Thus, the problem with using a uniform time
 197 profile when emissions are actually decreasing over time, lies not so much in how emissions
 198 are quantified over time *per se* (i.e. in physical terms) as in the discounting and pricing of
 199 these emissions over time. With *i*) the incorrect time distribution of LUC emissions used in
 200 the European energy policy and *ii*) the common use of CBA as a decision-making tool in the
 201 decision-making sphere, we address the issue of project appraisal distortion in the context of
 202 emissions induced by LUC.

203 3 Cost-benefit analysis and the time profile of LUC emissions

204 In this section, we apply the CBA approach to the two temporal profiles of LUC emissions
 205 and determine the direction of the bias (subsection 3.1) as well as its magnitude in the case
 206 of wheat-based ethanol in France (subsection 3.2). Because the dynamics of LUC is our main
 207 focus, the model exclusively represents the part of CBA that monetises LUC (carbon-related)
 208 impacts.¹⁷¹⁸

209 3.1 A two-period NPV model

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

215 Consider a project that releases emissions as a result of land conversion¹⁹ at $t = 0$. The
 216 carbon price grows at the carbon price growth rate denoted $g \in [0, 1]$ such that the carbon price
 217 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
 218 in the project $r \in [0, 1]$, the NPVs associated with the uniform and differentiated approaches are

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

¹⁶Providing 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.

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

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

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

219 such that, for all $z_0, z_1 \in \mathbb{R}^+$:

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

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

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

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

228 3.1.1 Discounting effect ($0 < r \leq 1$ and $g = 0$)

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

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

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

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

232 leading to Proposition 1.²⁰

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

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

²⁰The 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.

242 **3.1.2 Carbon price effect ($0 < g \leq 1$ and $r = 0$)**

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

244 The NPV difference is

$$\Delta NPV = \frac{1}{2}p_0g(z_1 - z_0) < 0, \quad (5)$$

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

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

246 leading to Proposition 2.²¹

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

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

257 **3.1.3 Combined effect ($0 < r \leq 1$ and $0 < g \leq 1$)**

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

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

267 When the discounting and carbon price effects perfectly cancel each other out, the uniform
 268 and differentiated time distributions are strictly equivalent within CBA (i.e. the same NPV).

²¹The 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.

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

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

286 3.2 Numerical illustration: the case of French wheat-based ethanol

287 France is the biggest bioethanol producer in Europe (USDA, 2018) and its production mainly
 288 relies on wheat (Ademe, I Care and Consult, Blézat consulting, CERFrance, Céréopa, 2017).
 289 In this subsection, we provide a numerical illustration of our theoretical results with the ex-
 290 ample of direct LUC engendered by wheat-based ethanol production in France. The analysis
 291 of direct LUC shows that most lands converted to cropland and in particular wheat cultivation
 292 are grassland (Chakir and Vermont, 2013; Poeplau and Don, 2013), which will therefore be
 293 the focus of our study.²² Direct LUC related to the conversion of grassland to wheat fields in
 294 Europe accounts for approximately 30% of total emissions from life cycle and LUC impacts
 295 of bioethanol and approximately half of total LUC emissions, i.e., including indirect LUC
 296 (Fritsche et al., 2010, Figures 1 and 2).

297 **Assumptions** France is located in a temperate region where the increasing demand for bioen-
 298 ergy is leading to increasing rates of LUC (Poeplau et al., 2011). We assume that *i*) in the dif-
 299 ferentiated approach, carbon dynamics in the soil follow an exponential decrease across time

²²Grassland 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 element of the development of biofuels in France.

300 in line with Poeplau et al. (2011), and that *ii*) biomass-related emissions are instantaneous.²³
 301 Since this paper is mainly addressed to European policy-makers, we use a 20-year time horizon
 302 for LUC emissions as assumed in the European RED. The discount rates we employ are con-
 303 stant²⁴ and range from 0 to 5% in the analysis, which is in line with the estimated values of the
 304 discount rate found in cost-benefit analyses of public projects and policies in Europe (Florio,
 305 2014, p.187). We consider three scenarios: a 0% discount rate as the baseline, a 3% discount
 306 rate as recommended by the European Commission (2014) in the EU funds framework²⁵ and a
 307 4.5% discount rate as recommended by Quinet (2013) and France Stratégie (2017) for the eval-
 308 uation of public investment projects in France. Finally, for the sake of clarity in this subsection,
 309 we consider carbon prices that grow at a constant rate²⁶ close to average growth rates that can
 310 be found in existing carbon price scenarios. We consider an initial price of 87€ in 2020 as
 311 recommended by Quinet (2019). The initial carbon price is kept constant across scenarios for
 312 the sake of comparability. Each scenario is characterised by a specific carbon price growth rate
 313 as follows:

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

322 Because the initial price is assumed to be the same across all scenarios, environmental objec-
 323 tives are considered increasingly constraining from Scenario O to Scenario C. In addition to
 324 Scenarios O, A, B and C, we consider the carbon price trajectory of the Quinet (2019) report,
 325 henceforth shadow price of carbon (SPC) scenario, the carbon price growth rate of which is not
 326 constant over time (see Table I). For the sake of comparison between Scenario SPC and Sce-
 327 narios O, A, B and C, the average annual growth rate of the carbon price in the SPC scenario is
 328 9.1% between 2020 and 2040, the period over which we consider the biofuel project.

²³Nonetheless, 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 B.

²⁴We discuss this assumption in Section 5.

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

²⁶This assumption might presently be restrictive since most existing carbon price scenarios, which we explore in Section 4 (see Table I), 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).

329 **Data** The computation of LUC emissions relies on the formal definitions of the uniform and
 330 differentiated approaches as described in Appendix B. To determine carbon stock changes in
 331 soil and vegetation, we rely on the guidelines provided by the European Commission (2010),
 332 which are based on IPCC (2006). Such a calculation requires knowledge about climatic region,
 333 soil type, agricultural management, agricultural practices (input level) and crop yields. The
 334 assumptions on these factors for our case study are described in Appendix C. Regarding the
 335 share of carbon that is converted into CO₂ emissions, we assume that 30% of the carbon stock
 336 in soil is converted into CO₂ (as in Anderson-Teixeira et al., 2009). This figure falls in the
 337 range given by the Winrock database (see Table 1 in Broch et al. (2013)) and is very close to
 338 the assumption of 25% made by Tyner et al. (2010). We assume that the reverse conversion is
 339 symmetric. Regarding the carbon stored in vegetation, we hypothesise that 90% is converted
 340 into emissions – a figure in line with the CARB policy in the United States.²⁷ An overview of
 341 the data used in the study, including sources, is provided in Appendix C.

342 **Computation tool** We develop a Python program²⁸ to generate the uniform and differentiated
 343 time distributions and calculate the NPV of the GHG emissions of bioethanol projects under
 344 the two time profiles. Once LUC emissions due to soil and biomass carbon stock changes as
 345 well as their dynamics over time are determined,²⁹ carbon releases are converted into CO₂
 346 emissions according to Appendix B, and finally priced using one of the scenarios listed above.
 347 Regarding price scenarios, an algorithm extrapolates prices in an exponential way between two
 348 one-time carbon prices, which allows us to generate a complete trajectory of carbon prices over
 349 the time horizon considered, since only sparse carbon prices are provided in most scenarios,
 350 including the World Energy Outlook's (IEA, 2018, p.604). The program essentially returns all
 351 the environmental NPV types necessary for the analysis, i.e. types related to LUC emissions
 352 (under each type of time distribution), non-LUC emissions and total emissions from biofuel
 353 production (i.e. LUC + non-LUC).

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

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

²⁸Namely PyLUCCBA. The program (complete tool coded in Python language) is publicly available on GitHub, <https://github.com/lfauchoux/PyLUCCBA> and described in the supplementary material linked to this paper.

²⁹Referring to Appendix B 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).

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

359 ➤ **Discounting effect**

360 Figure 2 illustrates the discounting effect for grassland converted to cropland. Carbon
 361 prices are constant over time and equal to 87€/tonne of CO₂.

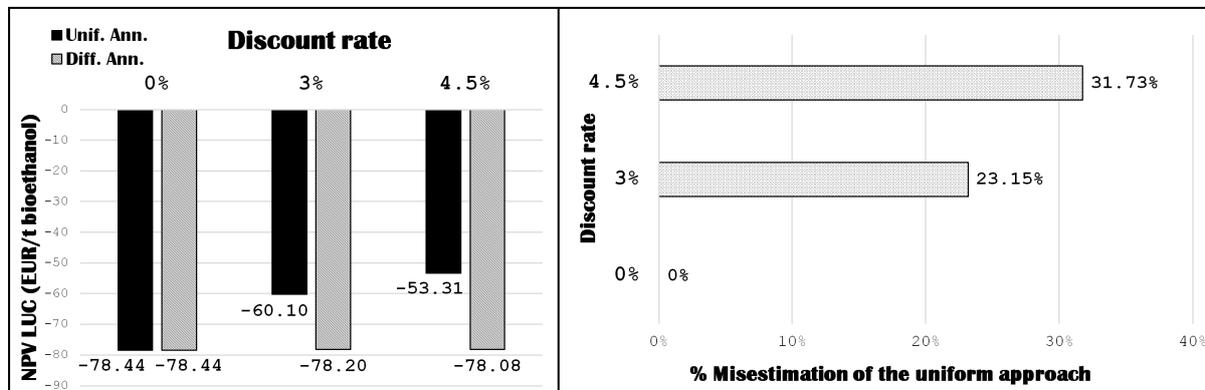


Figure 2: Net Present Value of LUC emissions (left) and relative upward bias induced by the uniform approach (right) for different discount rate values. For grassland conversion.

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

371 ➤ **Carbon price effect**

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

377 Figure 3 shows that the NPV of emissions due to LUC is underestimated under the uni-
 378 form approach (drops from -78.44€ to -223.02€). The higher the carbon price growth
 379 rate (from Scenario O to Scenario SPC), the larger the bias induced by the uniform ap-
 380 proach (downward bias ranging from 33.93% under Scenario A to 180.97% under Sce-
 381 nario SPC).

382 ➤ **Combined effect**

383 When combining a positive discount rate (fixed to 4.5% in line with evaluations of public
 384 investment projects in France) with an (average) carbon price growth rate ranging from

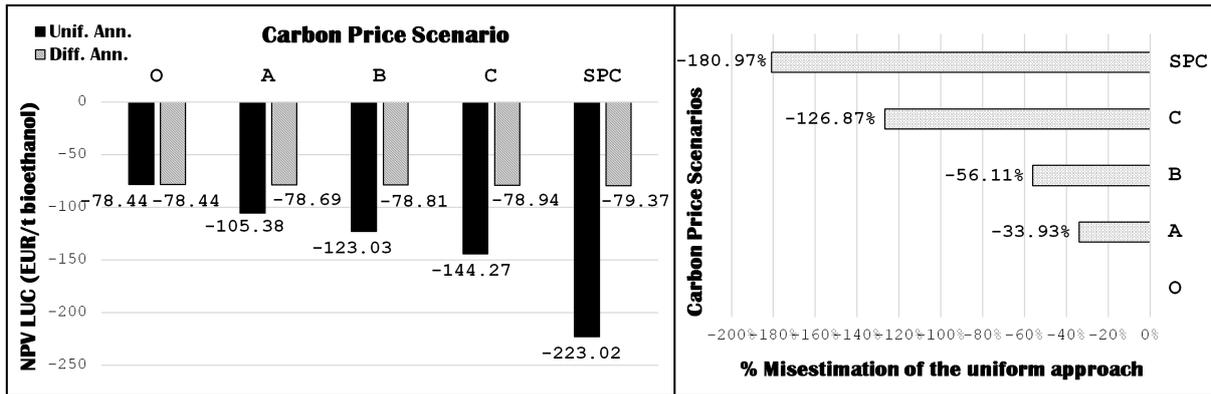


Figure 3: Net Present Value of LUC emissions (left) and relative downward bias induced by the uniform approach (right) for different carbon price scenarios. For grassland conversion.

385 0% (Scenario O) to 9.1% (Scenario SPC), the direction of the bias depends on whether
 386 the carbon price growth rate grows faster or slower than the discount rate (see Figure 4).

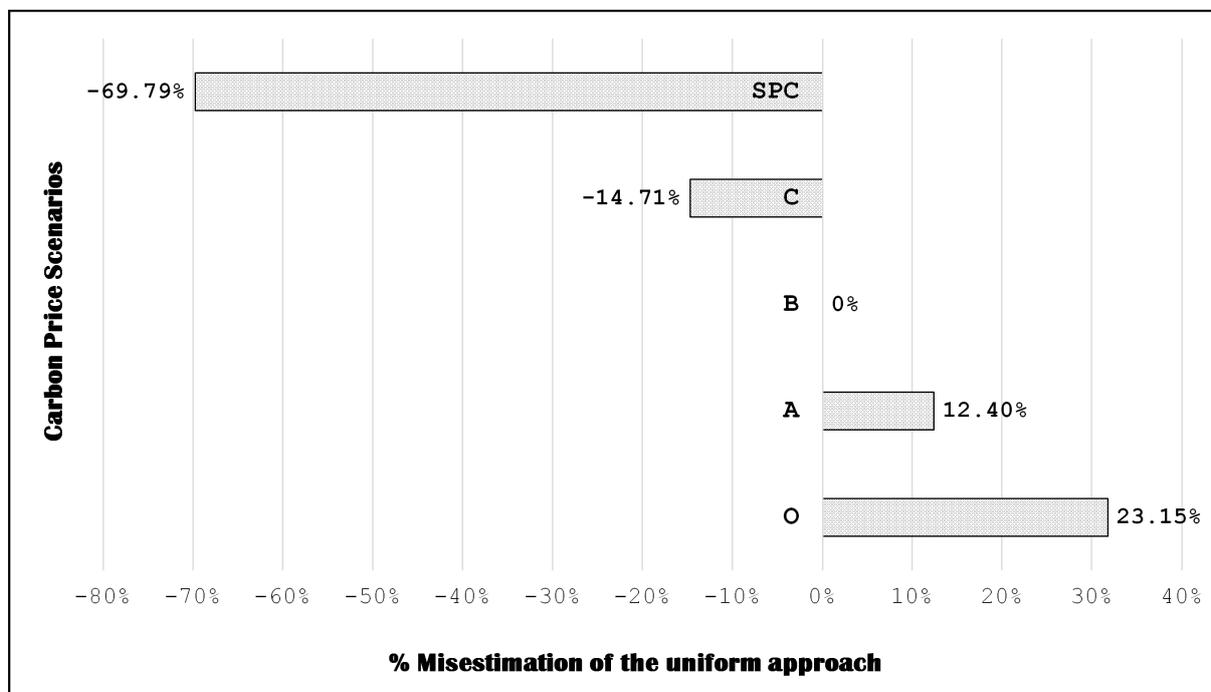


Figure 4: Relative bias induced by the uniform approach (4.5% discount rate and different carbon price scenarios). For grassland conversion.

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

395 respectively.

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

400 4 Proposal of two simple tools for decision-makers

401 Given the NPV misestimation that the uniform approach induces, we provide two simple tools
402 to help decide whether to implement a biofuel project, namely the compensatory rate (sub-
403 section 4.1) and the carbon profitability (CP) payback period (subsection 4.2). Our tools ex-
404 clusively rely on the environmental, i.e. non-market-related, part of CBA for several reasons.
405 First, because CBA is monetary *per se* and thus aggregates monetised environmental flows with
406 market flows, the economic NPV, i.e. market-related,³¹ would just be translated by the environ-
407 mental NPV downwards (upwards) in the case of net emissions (sequestrations) related to the
408 project. Therefore, the environmental NPV, calculated by the Python program available online,
409 can simply be added to the economic NPV. Second, the economic part of CBA relies on mul-
410 tiple (private) determinants such as land prices, competitive advantage and political context.
411 By contrast, the environmental part of CBA is independent of the project holder's specificities
412 and relies on isotropic determinants such as the conversion rate of carbon fluxes into carbon
413 emissions and standard carbon price trajectories.³² The particularities of the environmental
414 part of CBA are all incorporated in the Python program developed for the purpose of this study
415 and, more generally, decision-making. All specificities can be changed or enriched³³ accord-
416 ing to the project at hand, e.g. crop type and its consequences on carbon stock changes and
417 emissions from cultivation and production processes. Third, the capacity of the atmosphere to
418 handle GHG emissions is limited, which makes the consideration of the environmental part of
419 CBA interesting. The traditional use of payback periods of a project in economic calculation is
420 informative, but we argue that it could be complemented with carbon-specific payback periods
421 as presented in subsection 4.2, if one wishes to emphasise environmental concerns in the CBA
422 context.

423 4.1 Compensatory rate

424 We define the compensatory rate as the discount rate value that cancels the bias induced by the
425 uniform approach given a carbon price path. Put differently, it is the rate that equalises NPVs

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

³²Often specific to a whole region or country.

³³Indeed, 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.

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 I.

Table I: 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	

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

The compensatory rate³⁶ 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

³⁴If 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.

³⁵The 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.

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

446 use of the uniform time distribution given a specific carbon price trajectory. Figure 5 provides
 447 a numerical illustration of the compensatory rate applied to the carbon price trajectories de-
 448 scribed in Table I in the context of bioethanol production in France (related to the conversion
 449 of grassland to cropland). The more constraining the scenario, the higher the compensatory
 rate.

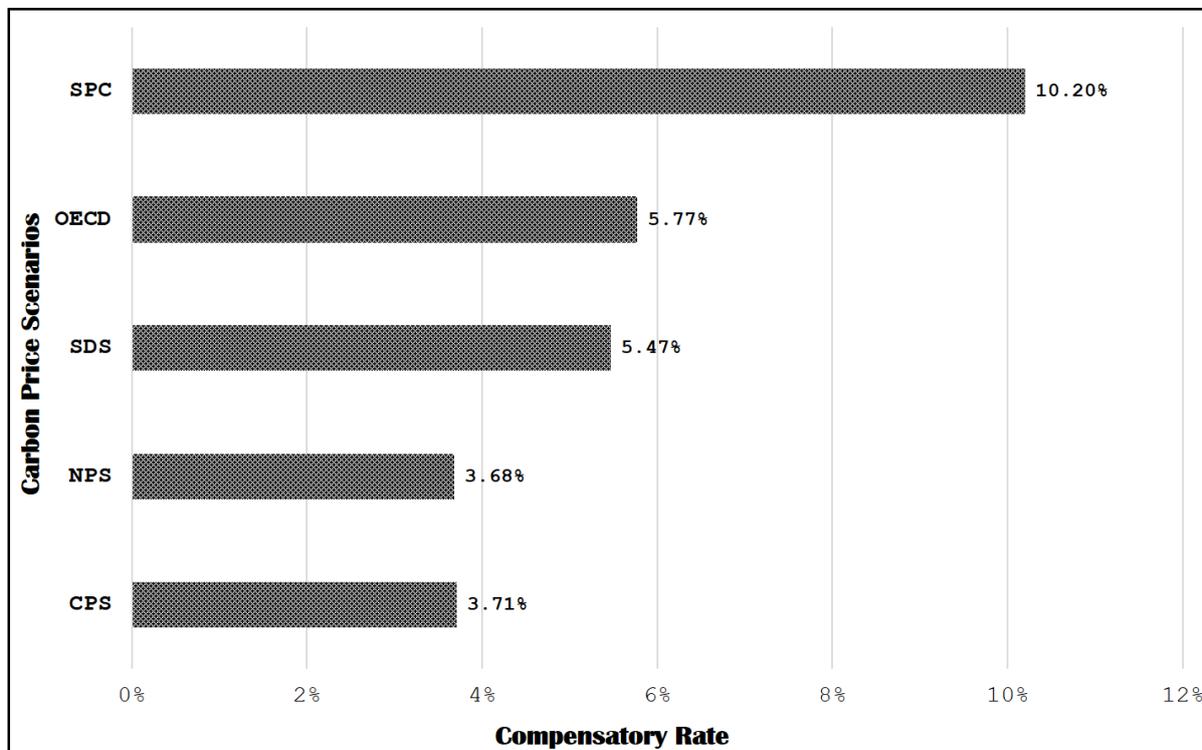


Figure 5: Compensatory rate across different carbon price scenarios, conversion of grassland to cropland.

450

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

457 4.2 Carbon profitability (CP) payback period

458 The second tool to help decide whether to implement a biofuel project relies on the whole envi-
 459 ronmental part of CBA, i.e. on including LUC and non-LUC emissions. Non-LUC emissions
 460 encompass emissions from the production, transport and distribution of biofuels and the culti-
 461 vation of energy crops. As in Section 3, we consider land conversion from grassland to wheat
 462 fields. Bioethanol projects are compared with fossil fuel production projects based on equiv-
 463 alent amounts of energy produced. In this context, GHG savings are allowed because aside

464 from LUC emissions, the amount of GHGs emitted from the production and consumption of
 465 fossil fuels is greater than the energy-equivalent GHG amount from bioethanol production and
 466 consumption.

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

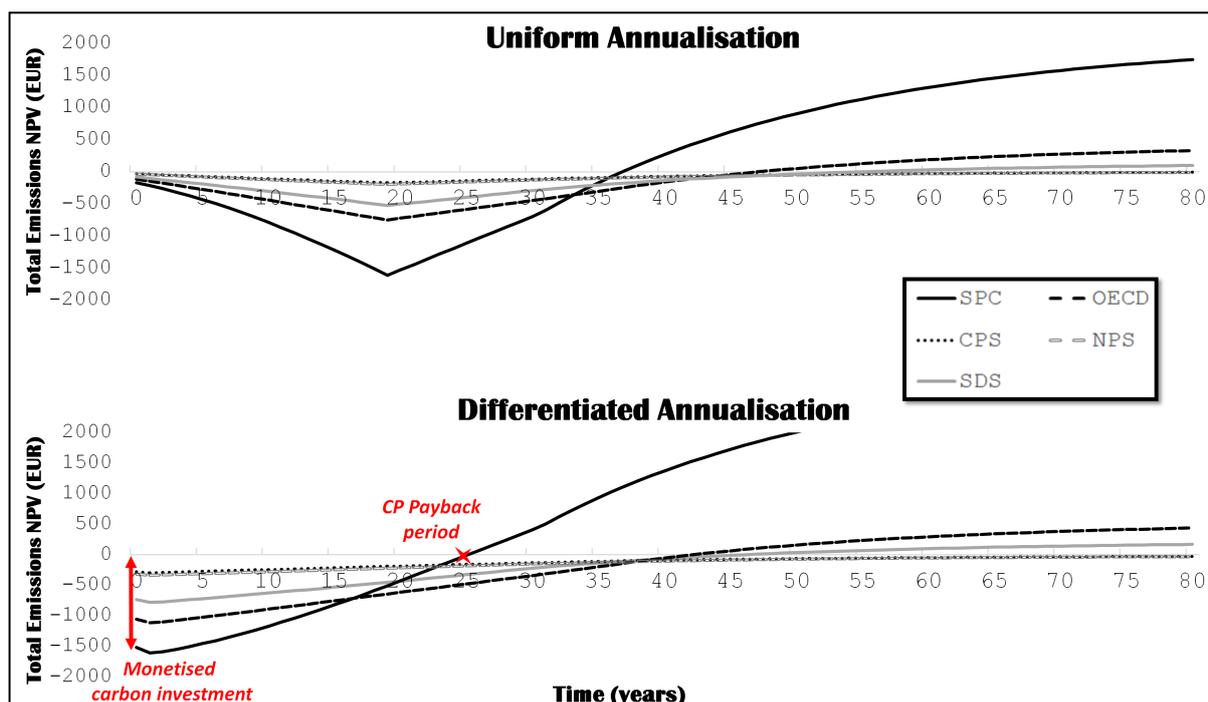


Figure 6: Carbon profitability payback periods across different carbon price scenarios under the uniform (top chart) and the differentiated (bottom chart) time distribution.

479

480 The CP payback period changes across scenarios and across time distributions as reported
 481 in Table II.³⁸ Overall, all payback periods are relatively high (higher than the time horizon of

³⁷Note 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 $t = 1$.

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

482 the project). The payback periods computed under the uniform approach for the SPC, OECD
 483 and SDS scenarios are greater than those under the differentiated approach. By contrast, the
 484 payback periods computed under the uniform approach for the CPS and NPS scenarios are
 485 smaller than those under the differentiated approach.

Table II: 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

486 The problem with using the uniform approach is that an LUC-related project may pass the
 487 CBA test under the differentiated approach but not under the uniform approach or vice-versa.
 488 If decision-makers use a benchmark CP payback period, which should be pre-established by
 489 policy-makers,³⁹ this benchmark could be compared to the CP payback period of projects. E.g.
 490 in the SPC scenario, if the benchmark were fixed to 30 years, the project would not pass under
 491 the uniform approach while in reality (i.e. under the differentiated approach), emissions do
 492 comply with such a requirement, thereby penalising projects that would actually be considered
 493 as beneficial to the environment according to predetermined benchmark. By contrast, with
 494 the NPS scenario, where the carbon price grows slower than the discount rate, the uniform
 495 approach may end up lending support to projects that are actually harmful to the environment.⁴⁰
 496 Therefore, the CP payback period addresses the issue of decision error when mainly or partly
 497 based on CBA. The uniform approach may either be at odds with the primary objective of
 498 cutting emissions by not rejecting environmentally harmful projects or lead to the disapproval
 499 of projects that actually comply with the requirements (e.g. the benchmark payback period).

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

505 A limitation of this tool may be the absence of consideration of potential scale effects in
 506 biofuel production. Indeed, the carbon profitability payback period also involves non-LUC
 507 emissions from the production process, and thus, it is subject to economies of scale (which

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

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

508 is not the case for LUC emissions). Intuitively, taking these economies of scale into account
509 would shorten the estimated payback periods for both time distributions since economies of
510 scale lead to higher energy efficiency in biofuel production and thus faster net GHG savings
511 across the whole project time horizon. Nevertheless, nothing would change the aforementioned
512 conclusions regarding the comparison between the uniform and the differentiated approach.

513 5 Discussion

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

517 **The CBA framework** Cost-benefit analysis is a decision-support tool that is widely used in
518 project evaluation (OECD, 2018b). Its popularity can partly be attributed to its convenience
519 and simplicity in aggregating market flows with priced (non-market) CO₂ flows, resulting in
520 a synthetic assessment indicator, i.e. the NPV of the project. Nonetheless, we do not argue
521 that it is the only approach that should be used in project assessments. Instead, we emphasise
522 that such a widespread tool, whose influence on final decisions varies from moderate to large
523 (OECD, 2018b), should be used with caution when environmental impacts are characterised
524 by a peculiar time profile like LUC. Cost-benefit analysis should not be considered a unique
525 answer to project assessment, especially when other environmental impacts (on e.g. biodiver-
526 sity or water, the monetary valuation of which may not exist or may not be as robust as carbon
527 values) are affected by the project (OECD, 2018b, Figure 16.9). This economic tool should
528 be complemented with other approaches such as multicriteria analysis that can account for di-
529 mensions beyond e.g., economic efficiency (OECD, 2018a). Overall, "the role of CBA remains
530 one of explaining how a decision should look if the economic approach is adopted." (OECD,
531 2018c). This paper aims at promoting tools that, although economic, try to be consistent with
532 biophysical reality. Still, greater consideration should be given to the interdisciplinarity of ap-
533 proaches because it allows for a broader picture of the consequences of the implementation of
534 a project.

535 **Discounting and time horizon assumptions** Exponential discounting was assumed, in line
536 with the practice of project assessment guidelines suggested in European policies, i.e. within
537 a 20-year time horizon. While our objective was to raise the issue of not considering correct
538 land use change dynamics in *current practices* of socio-economic analysis, both assumptions
539 on the discount rate and the time horizon are worth discussing. First, such a short time horizon
540 is generally chosen to fit the expected duration of biofuel production. It has the advantage of
541 emphasising the importance of large upfront emissions due to land conversion. Yet, it does
542 not account for (i) the persistence of GHGs in the atmosphere for long periods, (ii) the future

543 of energy cropland (e.g., land reversion) and (iii) intergenerational issues. In a way, using a
 544 short time horizon is a ‘conservative’ approach since longer time horizons come with growing
 545 uncertainty (Broch and Hoekman, 2012). Besides, there is a large debate around the value and
 546 trajectory of the discount rate over time. While some economists are in favour of discounting
 547 environmental values, others are more reluctant to the idea. Still, economists agree on the need
 548 to reconcile discounting with sustainability and intergenerational equity (Martínez-Paz et al.,
 549 2016). Discounting relies on two main arguments: (i) individuals have a pure preference for the
 550 present and (ii) future generations are expected to be richer than today, increasing consumption
 551 inequalities over time (Gollier, 2002). No or low discounting gives more weight to the well-
 552 being of future generations. Within CBA, the objective is to apply the Hotelling rule to prevent
 553 discounting from overwhelming the value of emissions in the future. However, the rule is cur-
 554 rently hardly applicable because of the gap between current carbon prices and those that should
 555 reflect objectives of global warming limits (Quinet, 2019). Despite the lack of consensus on
 556 the suitable value for discount rates, the use of declining discount rates, as introduced in France
 557 by the Lebègue (2005) report for public investment projects, has become more common under
 558 longer time horizons (Guesnerie, 2017). This allows one to put less weight on the longer term,
 559 which is characterised by uncertainty surrounding both economic growth and long-term envi-
 560 ronmental impacts (see e.g., Arrow et al., 2013; Arrow et al., 2014). In France, the declining
 561 profile of discount rates is effective only 30 years after the project starts, which we did not ex-
 562 plore in our numerical illustration as it considers 20 years as the time period over which ethanol
 563 production and LUC impacts should be examined (IPCC, 2006; European Commission, 2010;
 564 Delucchi, 2011). Nonetheless, when using the CP payback period, a benchmark in excess of
 565 30 years would justify the use of declining discount rates in our calculations.

566 **Extension to indirect land use change** Because of the uncertainty surrounding the identi-
 567 fication and quantification of indirect LUC (Di Lucia et al., 2012), we only dealt with direct
 568 LUC. However, the philosophy behind the model can apply to any phenomenon that entails
 569 the same carbon dynamics, thereby including indirect LUC.⁴¹ It is worth emphasising that the
 570 magnitude of the bias can be expected to increase with the accounting of indirect LUC, which is
 571 currently a central issue in European policies (European Commission, 2015a; European Com-
 572 mission, 2018b). Besides uncertainty, indirect LUC differs from direct LUC in terms of the
 573 stage of a project at which it arises. Indeed, Zilberman et al. (2013) point out that indirect
 574 LUC occurs with significant time lags. Empirical evidence suggests that the materialisation of
 575 indirect LUC takes 10-15 years after land is converted to energy crop fields (Andrade De Sá
 576 et al., 2013). This implies that, with a 20-year time horizon, a potentially large part of indirect
 577 LUC emissions related to a project would be truncated in CBA. Indeed, like direct LUC emis-
 578 sions, indirect LUC emissions should be considered over a 20-year time period as suggested by

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

579 IPCC reports. The application of the uniform approach would strongly affect the accounting of
 580 indirect LUC emissions because all emissions above the time horizon (i.e. constant emissions
 581 over 5-10 years) would not be considered. If instead, the differentiated approach applies to in-
 582 direct LUC emissions, most emissions would be accounted for within the period over which the
 583 project is considered. Therefore, the use of the uniform approach for indirect LUC emissions
 584 would enhance the misestimation of the NPV for two reasons: (i) the larger truncation of emis-
 585 sions under the uniform approach than under the differentiated approach and, (ii) the fact that
 586 emissions under the uniform approach undergo the discounting and carbon price effects more
 587 than under the differentiated approach. If one wishes to consider the entirety of carbon-related
 588 impacts of a biofuel project, an adaptation of the time period over which biofuel production
 589 projects are assessed is necessary when indirect LUC emissions are considered in CBA.

590 **Second-generation biofuels and carbon sequestrations** While the focus of our paper was
 591 LUC emissions, our framework could also apply to LUC sequestrations.⁴² Second-generation
 592 biofuel projects are particularly promising for carbon sequestration (see e.g. Anderson-Teixeira
 593 et al., 2009; Nakajima et al., 2018) conditional on energy crops not replacing lands with higher
 594 carbon content (Don et al., 2012). There are a growing number of second-generation biofuel
 595 projects in France, e.g. Futurol and BioTfuel. However, the dynamics of LUC sequestrations
 596 are less clear than those of LUC emissions. The results of the meta-analysis by Qin et al.
 597 (2016) and the study by Poeplau and Don (2014) suggest that sequestrations are not constant
 598 over time and might not even be monotonic, thereby questioning again the uniform time dis-
 599 tribution assumption currently adopted in European policy.⁴³ Provided the knowledge of the
 600 correct time distribution of sequestrations, only the compensatory rate would be useful to sup-
 601 port decisions. Indeed, sequestrations constitute a benefit to society, which makes the use of
 602 CP payback periods irrelevant.

603 **LUC dynamics and carbon markets** Reductions of emissions from LUC are part of the
 604 2018 European Union climate legislation for the period 2021-2030 (European Commission,
 605 2018b). Although LUC considerations are not covered by the European Emission Trading
 606 Scheme (ETS) (Hamrick and Gallant, 2017; ICAP, 2019),⁴⁴ the implications of the discrepancy
 607 between LUC temporal profiles under this widespread quantity-based instrument are worth dis-
 608 cussing. Currently, the sectors covered by the EU ETS, e.g. energy, purchase permits in line
 609 with their effective annual needs. This would not be the case under the uniform approach that

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

⁴³The 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.

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

610 does not reflect the real dynamics of LUC emissions. If LUC emissions were capped, the
611 consideration of the uniform approach would allow biofuel producers to smooth their need for
612 emission permits over time. However, biofuel producers would also suffer from increasing
613 prices over time. If instead the differentiated temporal profile were adopted, biofuel producers
614 would not be able to smooth their need for carbon allowances over time. They would most
615 likely need to purchase permits in the early phase of production, the upfront purchase poten-
616 tially weighing heavily in their cost-benefit balance depending on carbon market prices.

617 6 Concluding remarks and policy recommendations

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

633 The objective of this paper was to raise the *current* accounting for LUC dynamics in Euro-
634 pean policy and the problem it might cause in project assessment when CBA is used. Economic
635 processes, reflected in CBA, treat different points in time differently through the use of dis-
636 count rates and increasing carbon prices whereas policy assumptions, often based on life-cycle
637 assessment results, uniformly amortise LUC emissions over time. Our first-best recommen-
638 dation, specifically addressed to policy-makers, is to correct for this disconnection in policy
639 assumptions by relying more on academic research on the dynamics of LUC. This would avoid
640 misleading NPV results when CBA is used as a decision-support tool. If the available empir-
641 ical evidence (e.g. Poeplau et al., 2011) is deemed insufficient, a reasonable alternative that is
642 closer to the biophysical reality than the uniform approach would be to consider that the to-
643 tal emissions from biomass removal in connection with land conversion are felt immediately
644 instead of spread evenly over time. It is worth mentioning that the US biofuel policy (RFS2)

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

645 has gone a step forward (compared with the European Union) by disentangling the two carbon
 646 sinks (soil and biomass): biomass-related LUC emissions are fully accounted for at the time
 647 of land conversion while soil-related LUC emissions are uniformly distributed over time. A
 648 broader classification of the time distribution approaches used by policy-makers is provided in
 649 Appendix B. Still, since the recent Renewable Energy Directive reiterated the uniform time dis-
 650 tribution assumption (European Commission, 2018a), we recommend the use of the two tools
 651 suggested in this paper in the context of project assessment to complement traditional CBA re-
 652 sults.⁴⁶ The compensatory rate and the carbon profitability payback period are provided by the
 653 online Python program once a project of interest has been specified. The program allows (pub-
 654 lic or private) decision-makers to obtain the environmental part of their project's NPV, which
 655 can easily be added to the economic part. Both the compensatory rate and the carbon profitabil-
 656 ity payback period are adapted to the current policy situation and are therefore necessary while
 657 waiting for the transition towards more consideration of LUC dynamics in policy.

658 Appendices

659 A Proof of Proposition 3

$$\Delta NPV = NPV_u - NPV_d \quad (\text{A.1})$$

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

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

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

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

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

661 B LUC emissions time distribution: formal description

662 The following formal definitions of the uniform and differentiated approaches are implemented
 663 in the Python program to generate the numerical results provided in subsection 3.2 and Section
 664 4.

665 Let us denote by *SOC* and *VGC* the carbon stocks in soil and vegetation (biomass), re-

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

666 spectively, expressed in tonnes of carbon per hectare. Then, $\Delta SOC = SOC_F - SOC_I$ and
 667 $\Delta VGC = VGC_F - VGC_I$ are the carbon stock differences between land conversion and equilib-
 668 rium achievement where I and F refer to initial (before conversion) and final (after conversion)
 669 lands, respectively. z_t is expressed in tonnes of CO₂ per unit, e.g. hectare or tonne of ethanol,
 670 per year. z_t is decomposed into z_t^s and z_t^v the *annual* LUC emission flow from soil and vegeta-
 671 tion, respectively. z_t^s and z_t^v are respectively spread out over the time horizons T^s and T^v . ω_s
 672 and ω_v are introduced as the respective shares of soil and vegetation carbon that are converted
 673 into CO₂ emissions.⁴⁷ A is a constant that includes at least the coefficient of conversion of
 674 carbon into CO₂.⁴⁸

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

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

678 with $z_t^v = 0 \forall t \geq T^v$.

Definition 2 (differentiated annualisation) *LUC emission flows are “differentially” annu-*
alised when $T^v \leq T^s$, $z_t^s \neq z_{t+1}^s \forall t \leq T^s$ and $z_t^v \neq z_{t+1}^v \forall t \leq T^v$. Then, the total annualised
LUC emission is

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

679 with $z_t^v = 0 \forall t \geq T^v$.

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

682 For a grassland or a forestland converted into a cropland, SOC decreases exponentially accord-
 683 ing to the meta-analysis of Poepplau et al. (2011).⁴⁹

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

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

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

⁴⁹Such that $f^s(t) = e^{-\frac{t-1}{a}} - e^{-\frac{t}{a}}$ where a is a constant. Poepplau et al. (2011) estimate stock dynamics such
 that $\forall t, SOC_t = \Delta SOC(1 - \exp(-\frac{t}{a}))$. 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 III: Weak and Strong Definitions of LUC time distributions

		<i>Time Horizons</i>	
		$T^v < T^s$	$T^v = T^s$
<i>Carbon Dynamics</i>	No	Weak Uniform	Strong Uniform
	Yes	Strong Differentiated	Weak Differentiated

688 Definition 3 allows us to categorise energy policies according to the time distribution they
689 consider for LUC emissions. The uniform annualisation definition is strong in the sense that
690 it is the extreme case of uniformisation: emission flows (from both soil and vegetation) are
691 equal over the same time period. This is a far cry from the real dynamics of LUC. By contrast,
692 the differentiated annualisation definition is strong in the sense that soil- and vegetation-related
693 LUC emissions are distinguished in both their time horizon and their dynamics. The strong dif-
694 ferentiated annualisation is the closest definition to what is described in the scientific literature.
695 The European RED is based on the strong uniform annualisation definition with the assump-
696 tion that $T^v = T^s = 20$, and the U.S. RFS2 policy is based on the weak uniform approach with
697 $T^v = 1$ and $T^s = 30$.

698 C Data

Table IV: Data Used for the Bioethanol Case Study in France

About	Choice/Value	Reference
Region	France	-
Biofuel	Bioethanol	-
Biomass 1 st genera- tion	Wheat	Chakir and Vermont (2013)
Project Starting Year	2020	-
Discount rates	From 0% to 5%	Florio (2014) and Quinet (2013)
Project Time Horizon	20, $t = 0$ land conversion Period of production: 20 yrs from $t = 1$ to $t = 20$	European Com- mission (2009a), European Commis- sion (2015b), and European Commis- sion (2018a)
Carbon Price Projec- tions	WEO trajectories, OECD ques- tionnaire, Shadow price of carbon in France	IEA (2018), OECD (2018b), and Quinet (2019)
Crop Yields	Wheat: 7.5 t DM/ha <i>Miscanthus</i> : 16.5 t DM/ha	Agreste IFP energies nou- velles

Process Yields	Wheat: 0.28 t eth/t DM <i>Miscanthus</i> : 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, <i>Miscanthus</i> , Improved Grassland, Degraded Grassland, Forest	-
Agricultural Management	Wheat: 60% Full tillage & 40% No till <i>Miscanthus</i> : No till	Agreste
Agricultural Practices	Wheat: 70% High input without manure 30% with manure <i>Miscanthus</i> : Medium Input	Agreste
Coefficient shares carbon to CO2	Emi: $\omega_s = 30\%$ and $\omega_v = 90\%$ Seq: $\omega_s = 30\%$ and $\omega_v = 100\%$	See subsection 3.2 of the paper
Non-LUC emissions	Wheat <i>Miscanthus</i>	Biograce Hoefnagels et al. (2010)
Gasoline emissions	87.1 g CO2/MJ	Joint Research Centre (JRC WTT report Appendix 2 version 4a, April 2014)

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