Biofuel policies and the environment: the effects of biofuel feedstock production on climate, water quality and biodiversity

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Abstract:
In this paper we examine the multiple environmental effects of policies promoting biofuel production from agricultural crops. We develop theoretical and empirical frameworks and provide an integrated economic and ecological modelling approach: an economic model of farmers' decision making is combined with a biophysical model predicting the effects of farming practices on crop yields and multiple environmental effects. The analysed environmental effects include GHG emissions over the life cycle, nitrogen and phosphorus runoff, herbicide runoff and the quality of wildlife habitats. Model is applied to crop production in Finland. We found that the overall environmental performance of alternative land use types is mainly driven by the value of CO_2-eq emissions and nutrient runoff damage. Herbicide use intensity and resulting herbicide runoff damage have only a marginal effect on the environmental performance of alternative land use types. Incorporation of biodiversity benefits favour rape and reed canary grass over cereals. Social welfare ranking of alternative land use types is mainly driven by profitability of land use rather than the social valuation of environmental effects.

Keywords: life cycle analysis, nutrient runoff, herbicide runoff, greenhouse gas emissions

1. Introduction

A comprehensive analysis of bioenergy policies should take into account all relevant environmental effects (soil, water, air, biodiversity, landscape and climate) over the life cycle. To date, the bulk of bionergy and biofuels literature has focused on net energy balances and net greenhouse gas (GHG) emissions of alternative bioenergy and biofuels options. The literature on other environmental effects of biofuel production, such as effects on water quality and biodiversity, is almost non-existent.

Governments' biofuel support policies include budgetary policies (tax concessions, tax credits and direct support to biofuel processing industry), biofuel mandates (blending equirements) and import tariffs. In this paper we examine multiple environmental impacts of alternative scenarios related to biofuel support policies. We provide an integrated economic and ecological modelling approach: an economic model of farmers' decision making is combined with a biophysical model predicting the effects of farming practices on crop yields and multiple environmental effects. The analysed environmental effects include GHG emissions over the life cycle, nitrogen and phosphorus runoff, herbicide runoff and the quality of wildlife habitats.

Following OECD (2008) we consider three alternatives including the current policy baseline. In the baseline the growth in the production and use of both ethanol and biodiesel continues due to existing policies supporting biofuel production and use at different stages of the marketing chain. Secondly we investigate the effects of two recently announced and enacted biofuel programs. These programs are the US Energy Independence and Security Act (EISA) enacted in December 2007, and the new EU Directive on Renewable Energy (DRE) currently in the legislative process. While the former defines a new Renewable Fuel Standard calling for US biofuel use to grow to a minimum of 136 billion litres per year by 2022, the latter suggests replacing 10% of fossil fuels. Thirdly, given the recent criticism of biofuel support policies and doubts on their impacts on agricultural prices we examine the environmental effects in the case where these biofuel support policies are eliminated altogether. The model is applied to the Finnish agriculture.

The rest of the paper is organized as follows. Section 2 develops theoretical framework for the paper while empirical specification of model is presented in section 3. Finally, results and discussion are presented in section 4.

2. Theoretical framework

Consider biofuel (ethanol or biodiesel) processing firm. It combines bionergy crops (y) and energy (e) in the production process to manufacture biofuels. Let the production technology of the industry define a continous and concave production function of biofuel (h): $h = g(y, e)$ with $h_y > 0$, $h_e > 0$ but $h_{yy} < 0$, $h_{ee} < 0$. Let *q* denote the price of biofuels, *p* the price of bioenergy crop and *v* the price of energy. We describe the direct support as a subsidy (s) to the use of bioenergy crops, thus ($p - s$) is the after-support price of bioenergy crops. As is well-known from other literature, a blending requirement (*m*) tends to increase the price of biofuel, so that we can express biofuel price as $q = q(m)$, with $q' > 0$. Import tariff will have similar impact on price, thus m can be used to describe both the impact of blending requirement and tariff. Equipped with this notation, the economic problem of the biofuel firm if to choose the use of inputs so as to maximize its profits, that is

$$\underset{y,e}{Max}\ \pi^b = q(m)g(y,e) - (p-s)y - ve\,. \qquad (1)$$

The conventional first-order conditions

$$\pi_y^b = q(m)g_y - (p-s) = 0 \qquad (2a)$$

$$\pi_e^b = q(m)g_e - v = 0 \qquad (2b)$$

define the demand function for inputs. In particular, it holds for the demand for bioenergy crops, y^d, that $y^d = y^d(q(m), p, s, v)$. By differentiation we have that $y_m^d > 0$ and $y_s^d > 0$, so that biofuel policies increase demand for bioenergy crops. Furthermore, assuming bionergy crop and energy input are complements in the production process, we have $y_v^d < 0$. Hence, higher price of energy decreases demand for bioenergy crops.

The amount of arable land, *A*, is allocated between bioenergy crops and food/feed crops. To keep the analytical discussion transparent, we treat land as homogenous by its quality but assume that cultivation costs differ due to accessibility, distance, and other factors (in the empirical application we let the land quality differ.) Let $f^i(l)$ denote the response function of both crops and ϕ the price of fertilizers. We denote the share of land allocated to crop 1 as *a* and (*1-a*) to crop 2 (bioenergy crop). Then the land related, site-dependent cultivation costs are $c(aA)$ and $c((1-a)A)$, respectively. Both cost functions are increasing in land. The land areas with lower cultivation costs are allocated to the more profitable crop. The profit function of the farmer can now be expressed as

$$\Pi^F = \pi^1 aA + \pi^2 (1-a)A \qquad (3)$$

where $\pi^1 = \left[p_1 f^1(l) - \phi l_1 - c(aA)\right]$ and $\pi^2 = \left[p_2 f^2(l) - \phi l_2 - c((1-a)A)\right]$.

The farmers' economic problem is to choose fertlizer intensity for both crops and to allocate the land area between the two crops. The firs-order conditions are given by,

$$\Pi_{l_1}^F = pf_{l_1}^1 - \phi = 0\text{; and } \Pi_{l_2}^F = pf_{l_2}^2 - \phi = 0 \tag{4a}$$

$$\Pi_a^F = \pi^1 - c'(aA) - \pi^2 + c'((1-a)A) = 0 \tag{4b}$$

While the choice of fertilizer intensity for both crops is evident and independent of land allocation, the condition for land allocation is interesting. We can re-express it as $\pi^1 - c'(aA) = \pi^2 - c'((1-a)A)$, so that the optimal share of land allocated to crop 1, $a*$, is obtained at the point where the profits from both crops are equal. Thus, the farmer allocates the first parts of land to the crop having originally higher profits. Gradually cultivation costs increase making the other crop more competitive. Finally, the switching point $a*$ is reached which divides land area between crops.

Drawing on the first-order conditions, output supply functions for both crops can be obtained. In particular for bionergy crop we have $y^s = f^2(l_2^*(p,\phi))(1-a(c,p,\phi))$. For an increase in the price of bionergy crop and fertilizer price we obtain $y_p^s = f_l^2 l_p (1-a) - f(\cdot)a_p > 0$ and $y_\phi^s = f_l^2 l_\phi (1-a) - f(\cdot)a_\phi < 0$.

Finally, market equilibrium of bionergy crops is obtained as an equality of demand and supply, that is when $y^d(q(m), p, s, v) = y^s(p,\phi,c)$. It is evident, then, that the price of bioenergy crop becomes a function of bioenergy policies. Finally, market equilibrium and the properties of production technology in use determine the environmental impacts on climate, water quality and biodiversity:

$$\Omega^j = \Omega^j(l, a, y, ..), \tag{5}$$

where j = climate, water and biodiversity. We next turn to developing the empirical specification of our model.

3. Empirical specification of the model

We apply our model to data from the Uusimaa and Varsinais-Suomi provinces in Southern Finland. The cultivated agricultural land in the region was 475 400 hectares in 2006, which represents approximately 20% of cultivated land in Finland. The average farm size in 2006 was 42 ha. Agriculture in the region is predominantly crop production. The predominant soil type in this region is clay and the predominant tillage method is conventional tillage (mouldboard plough tillage). The most representative crops in 2006 were barley (27%), spring wheat (21%), oats (10%), and rape (10%) (Yearbook of Farm Statistics 2007). These four crops and set-aside (13%) are included as land use forms in the model; they represent 80% of total acreage in the region.

In addition to the above land use forms, we also include cultivation of reed canary grass (*Phalaris arundinacea* L.), because this is regarded as the most suitable bioenergy crop for the climatic conditions in Scandinavia (Landström et al. 1996). In the Finnish climatic and soil conditions, reed canary grass produces 6-8 tons of dry mass per hectare for a time period of 10-12 years (Pahkala et al. 2005).

3.1 Crop production and profits

We model per hectare crop yield as a function of nitrogen fertilization. Farmers use a compound fertilizer that contains nitrogen and phosphorus in fixed proportions and target yield response to nitrogen application. Mitscherlich yield function is applied for spring wheat, barley, and oats but quadratic yield function is used for rape:

$$y_i = \mu_i (1 - \sigma_i \, e^{-\nu_i N_i}) \quad \text{and} \qquad y_i = A_i + \chi_i N_i + \gamma_i N_i^2 \tag{6}$$

where y_i is yield per hectare, N_i is nitrogen use per hectare, and μ_i, σ_i and ν_i , as well as A_i, χ_i and γ_i are parameters. The former set of parameters are estimated by Bäckman *et al.* (1997) and latter estimates by Heikkilä (1980) on the basis of Finnish field experiments. These parameters were calibrated to match observed crop yields associated with known fertilizer application rates on soils of different productivity in Southern and South-Western Finland.

For the use of herbicides, we apply an exponential specification for the damage-abatement function (Lichtenberg and Zilberman 1986),

$$\Phi_i = 1 - \exp(-\eta_0 - \eta_1 x) \tag{7}$$

with $\eta_0 \geq 0, \eta_1 \geq 0$. Finnish experiments on herbicide treatments reveal that in comparison to untreated plots, MCPA applications increase crop yields on the average by 10% (MTT 2004). Thus, damage-abatement function parameters, η_0 and η_1 are calibrated for each crop to reflect these herbicide treatment results.

Farmers maximize their profits by choosing the optimal rates of fertilizer and herbicide application, and on the basis of profits obtained from alternative crops, allocate each differential productivity parcel to the highest profits use.

$$\pi^i = \hat{p}\mu(1-\sigma e^{-\nu N})(1-\exp(\eta_0 - \eta_1 x)) - cN - \alpha x - wh - \Theta - K + S \tag{8a}$$

$$\pi^i = \hat{p}(A + \chi N + \gamma N^2)(1-\exp(\eta_0 - \eta_1 x)) - cN - \alpha x - wh - \Theta - K + S \tag{8b}$$

Farmers' per hectare profits for spring wheat, barley, and oats are given by (4a) and per hectare profits for rape are given by (4b). Effective output price is given by $\hat{p} = p - \varphi - \omega$, where p is output price per kg, φ is grain drying costs per kg of output, and ω is transportation cost per kg of output. Because this application utilizes life cycle analysis data the transportation distance T is fixed at 200 km for fertilizer, 100 km for barley, spring wheat and oats, and 70 km for rape and reed canary grass. Following Flyktman and Paappanen (2005) the following cubic transportation cost function, ω, is applied for reed canary grass (transported as round bales): $\omega^i = \tau T^3 - \xi T^2 + \rho T + \vartheta$. For other crops, the Finnish data suggests a linear transportation cost: $\omega^i = \varsigma + \Lambda T$. Fertilizer price is denoted by c and herbicide price by α. Labour cost per hour is denoted by w and labor input by h. Θ denotes the other costs of cultivation per ha, which are fixed with respect to the chosen tillage method (conventional tillage) and include seed, fuel, lubricants etc. Capital costs, K refer to fixed costs of capital and include depreciation, interest, and maintenance. Finally, S denotes EU and national support payments (area payments) for crops.

The modelling of reed canary grass cultivation follows Lankoski and Ollikainen (2008). Unlike other modeled crops, reed canary grass is a perennial crop, which is planted for a 14 year production rotation and the annual harvests start from the third year. Fertilizer application is technologically fixed for the first two years. Farmers' profits from reed canary grass cultivation are given by

$$\pi^i = \sum_{t=3}^{n} \left[(1+r)^{-(t-1)} \left(p\mu(1-\sigma e^{-\nu l}) - cN - I - K + S \right) \right] - \overline{C}_1 - \overline{C}_2 (1+r)^{-1} . \qquad \text{(8c)}$$

In equation (4c), $\overline{C}_1 = E + K + S$ and $\overline{C}_2 = c\overline{N}_2 + I + K + S$ comprise the establishment and some other cost items during the first two years, 1 and 2. E is the establishment costs of reed canary grass comprising fuel and labour costs of primary tillage, secondary tillage, and herbicide application, as well as fertilizer, seed and herbicide costs. I denotes the variable costs of cultivation, K refers to fixed machinery costs and the annual crop area payments are S.

3.2 Environmental effects

Figure 1 in Appendix shows the environmental effects taken into account in the empirical application. We focus on three environmental topics: surface water quality, climate, and biodiversity.

Nutrient runoff

The modeling of nutrient and herbicide runoff follows closely Lankoski et al. (2006). We examine both nitrogen and phosphorus runoff and for phosphorus account for dissolved reactive phosphorus (DRP) and particulate phosphorus (PP). Because in compound fertilizer (NPK) the three main nutrients are in fixed proportions, nitrogen fertilizer intensity determines also the amount of phosphorus used.

The following nitrogen runoff function (Simmelsgaard 1991) is employed,

$$Z_N^i = \phi_i \exp(b_0 + bN_i), \tag{5}$$

where Z_N^i = nitrogen runoff at fertilizer intensity level N_i, kg/ha, ϕ_i = nitrogen runoff at average nitrogen use, $b_0 < 0$ and $b > 0$ are constants and N_i = nitrogen fertilization in relation to the normal fertilizer intensity for the crop, $0.5 \leq N \leq 1.5$. This runoff function represents nitrogen runoff generated by a nitrogen application rate of N_i per hectare and the parameter ϕ_i reflects differences in crops.

Drawing on Finnish experiments (e.g. Saarela et al. 1995) it is assumed that 1 kg increase in soil phosphorus reserve increases the soil P status (i.e., ammonium acetate-extractable P) by 0,01 mg/l soil. Uusitalo and Jansson (2002) estimated the following linear equation between soil P and the concentration of dissolved phosphorus (DRP) in runoff: *water soluble P in runoff (mg/l) = 0.021*soil_P (mg/l soil) – 0.015 (mg/l).* The surface runoff of potentially bioavailable particulate phosphorus is approximated from the rate of soil loss and the concentration of potentially bioavailable phosphorus in eroded soil material as follows: potentially bioavailable particulate phosphorus PP (mg/kg eroded soil) = 250 * ln [*soil_P* (mg/l soil)]-150 (Uusitalo 2004). Thus, the parametric description of surface phosphorus runoff is given by

$$Z_{DRP}^i = \varpi_i[\psi_i(0.021(\Phi + 0.01 * P_i) - 0.015]/100 \tag{6a}$$

$$Z_{PP}^i = \Delta_i[\zeta_i\{250\ln(\Phi + 0.01 * P_i) - 150\}] * 10^{-6} \tag{6b}$$

where ψ_i is runoff volume (mm), Φ is soil_*P* (common to all crops) and ζ is erosion kg/ha, and P_i is the phosphorus application rate. As in the case of nitrogen, the crop based differences in the runoff of dissolved and the potentially bioavailable particulate phosphorus are captured by parameters ϖ_i and Δ_i, respectively. *Soil_ P* is fixed at 10.6 mg/l, which is the average for Finnish FADN farms situated in southern and south-western Finland (Myyrä et al. 2005).

Herbicide runoff

As regards herbicide runoff, the focus is on MCPA (active ingredient). MCPA is degradable and therefore MCPA decay is calculated according to the equation $B_i e^{kt}$, where B_i denotes the amount of MCPA applied, t is the number of days after application, and the coefficient of degradation, k, is defined as $k = \frac{\ln 2}{DT50}$. Thus, the following degradation equation for MCPA is obtained,

$$MCPA = x_i * e^{-0.0693t} \qquad (7)$$

Equation (7) defines the amount of MCPA in the soil at each point of time.

MCPA runoffs are modeled using equation (8), adapted from Kreuger and Törnqvist (1998):

$$\log Z^i_{MCPA} = \left[1.1 + (1.1 * \log MCPA(kg/ha) + 0.00004 * Koc - 0.005 * DT50\right] \qquad (8)$$

where Koc is the soil sorption coefficient (normalized to soil organic carbon content) which is 125 in this application, DT50 is the soil half-life, 10 days for MCPA. Equation (8) is calibrated to reflect the MCPA runoff experiments on clay soils in Southern and South-Western Finland (Laitinen et al. 1996).

Greenhouse gas emissions

Greenhouse gas emissions are modelled on the basis of life cycle assessment (LCA) estimates provided by Mäkinen et al. (2006), who estimated CO_2-equivalent emissions for the whole chain from the production of inputs to the final use of bioenergy and biofuels. In this application the following aspects are included: (i) CO_2-eq emissions related to the transportation of crops, (ii) CO_2-eq emissions related to the manufacturing, transportation and application of fertilizers, herbicide, and lime (iii) CO_2 emissions from soil and (iv) CO_2-eq emissions from tillage practices, such as plowing, harrowing and planting as well as CO_2-eq emissions from harvest and grain drying.

With respect to the transportation of inputs and outputs the following assumptions are made. All transportation takes place with a EURO III class (capacity 60 tons) trailer truck (one-way 100% use of capacity). On the basis of this assumption the CO_2-eq emissions are 69.3 g/ton/km. The manufacture, transportation (200 km) and application (N_2O emissions from soil) of one ton of NPK (20-3-8) fertilizer produces 2.143 tons of CO_2-eq emissions, which translates into 10.715 kg CO_2-eq emissions per 1 kg of N fertilizer. The manufacture, transportation and application of one ton of pesticides causes 16.7 tons of CO_2-eq emissions and the corresponding figure for lime is 0.45 tons. The soil CO_2 emissions for spring wheat, barley, oats and rape are 1.43 t CO_2/ha/a, whereas reed canary grass and green set-aside sequester carbon by 0.05 t CO_2/ha/a.

Quality of wildlife habitat

Lehtonen et al. (2008) develop a wildlife habitat indicator, habitat quality index, which measures the impacts of land use and land management on both the quality and quantity of wildlife habitats. The habitat specific biodiversity weights or 'habitat quality indices' were derived from Finnish field surveys of vascular plants (Hyvönen and Salonen 2002; Ma et al. 2002; Hyvönen et al. 2003; Salonen et al. 2005) and butterflies (Kuussaari and Heliölä, 2004). These relative weights can be used directly when calculating a habitat index as a linear vector, multiplied by a land use vector including hectares under each crop or habitat type. Divided by total farmland, the resulting index represents the average biodiversity value of farmland in alternative policy scenarios. Relative weights for different land use types are as follows: green set-aside 529, reed canary grass 160, rape 200, and spring wheat, barley and oats 100.

3.3. Monetary values for environmental effects

Following Lankoski et al. (2006), we transform total P into N equivalents by multiplying total P by the Redfield ratio 7.2. The Redfield ratio describes the optimum N/P ratio for the growth of phytoplankton, relevant for algal growth in sea waters. The marginal damage from nitrogen equivalents is assumed constant, so that the damage function is given by

$$d(Z^i) = R_n(N_i + 7.2P_i)\,, \qquad (9)$$

where R_n is the constant social marginal damage. Drawing on Aakkula (1999) and Yrjölä and Kola (2004), Finnish consumers experience a damage value of 35 euros from the average per hectare agricultural nutrient runoff (13 kg/ha N and 2 kg/ha P).

For the herbicide runoff constant marginal damage is postulated, so that the damage function is given by

$$D(Z^i) = R_h Z_i^h\,, \qquad (10)$$

where Z_1^h consists of MCPA runoff. As for the value of R_h, herbicide runoff differs from nutrient runoff because of the toxic nature of herbicides. Siikamäki (1997) suggests the average WTP € 189.3 per kg of active ingredients in the case of a total abandonment of herbicide use in Finnish agriculture, and this estimate is used.

The climate benefits from biofuels are modelled as offset benefits from emissions of fossil fuels. The price of emission allowances is used as a proxy for the marginal climate damage (CO_2-eq emissions) and benefits (offsets). The estimates for the Kyoto trading period range between 10-30 euros and accordingly € 20/tonne is used as the estimate.

Marginal biodiversity benefits from butterfly species richness are assumed to be constant at € 57 per hectare, in accordance with the willingness to pay estimates obtained by Aakkula (1999).

4. Results and discussion

Results are presented for three scenarios: Baseline, Policy scenario 1 (Removal of biofuel support) and Policy scenario 2 (new EU and US biofuel legislation). Policy scenario 1 incorporates the projected average EU prices for wheat, barley, oats and rapeseed in 2013-2017. In this price scenario, all biofuel-related policy instruments are removed (budgetary support, mandates and tariffs) (for a details of this scenario see OECD 2008). Policy scenario 2 also incorporates the projected average EU prices for wheat, barley, oats and rapeseed in 2013-2017, but in this price scenario, the following policies and technology developments are taken into account: the US Energy Act, the EU Bioenergy Directive, and second generation biofuels (for a details of this scenario see OECD 2008).

Reed canary grass (RCG) represents second generation biodiesel, while rape represents first generation biodiesel, barley is used for ethanol, oats is used for feed, and wheat is the food crop.

For all scenarios the basic results regarding land allocation, input use intensity, production and profits are presented in Appendix Table 2. Detailed empirical results concerning the environmental effects of alternative crops and policy scenarios are presented in Table 1 below.

As Appendix Table 2 shows, in the Baseline, RCG is cultivated in the 2 lowest productivity parcels with low nitrogen use intensity. The low nitrogen application rate is due to the high unit transportation costs and thus a low effective output price for RCG. However, support payments and low production costs make it profitable to cultivate RCG in the lowest productivity parcels. Oats cultivation takes place in the second lowest land productivities with low nitrogen and herbicide use intensities. In comparison to the Baseline Policy scenario 1 shifts the land allocation towards oats and rape. Land allocated to RCG and wheat in the Baseline is now allocated to oats. Due to changes in price ratios and land allocation, the average nitrogen and herbicide application rate decreases for rape, while for oats both of these increases slightly, since oats cultivation shifts to higher land productivities. The Policy scenario 2 makes RCG cultivation again profitable and lowest productivity land is allocated to it. This policy scenario increases the profitability of wheat and rape cultivation, and thus these two crops exhaust the remaining land available for production and thus oats is not cultivated under this Policy scenario. The input use intensity increases for all of these crops relative to the Baseline.

Table 1 presents total environmental effects under the Baseline, Policy scenario 1 and Policy scenario 2.

Table 1. Baseline, Policy scenario 1, and Policy scenario 2: *total nitrogen runoff, total phosphorus runoff, total herbicide runoff, total CO_2-eq emissions*.

Crop	N-runoff, kg	P-runoff, kg	Herbicide runoff, kg	CO2-eq emissions, tons	Habitat index value
			Baseline		
RCG	9	1	-	1	
Oats	24	5	0.04	11	
Wheat	192	27	0.22	70	
Rape	106	19	0.17	43	
Total	332	52	0.42	125	138.6
			Policy scenario 1 – Removal of biofuel support		
Oats	148	30	0.23	66	
Rape	123	22	0.19	50	
Total	271	52	0.42	116	142.9
			Policy scenario 2 – New EU and US biofuel legislation		
RCG	19	3	-	2	
Wheat	166	23	0.19	60	
Rape	142	25	0.22	57	
Total	327	51	0.41	119	153.3

Relative to the Baseline total nitrogen runoff decreases in Policy scenario 1. This result is mainly driven by land allocation shift from fertilizer intensive wheat to the less fertilizer intensive crops oats and rape. Decreased input use intensity in Policy scenario 1 also results in a decrease of the total CO_2-eq emissions when compared to the Baseline.

In the Policy scenario 2, higher application rates of fertilizer and herbicide inputs for wheat and rape is offset by increased allocation of land to RCG, which is cultivated with low fertilizer intensity and no herbicide use. Decrease in CO_2-eq emissions is mainly driven by an increase in the land allocated to RCG, which has low fertilizer intensity and thus low CO_2-eq emissions. Moreover, unlike other crops RCG sequesters carbon and thus its CO_2 emissions for soil are negative.

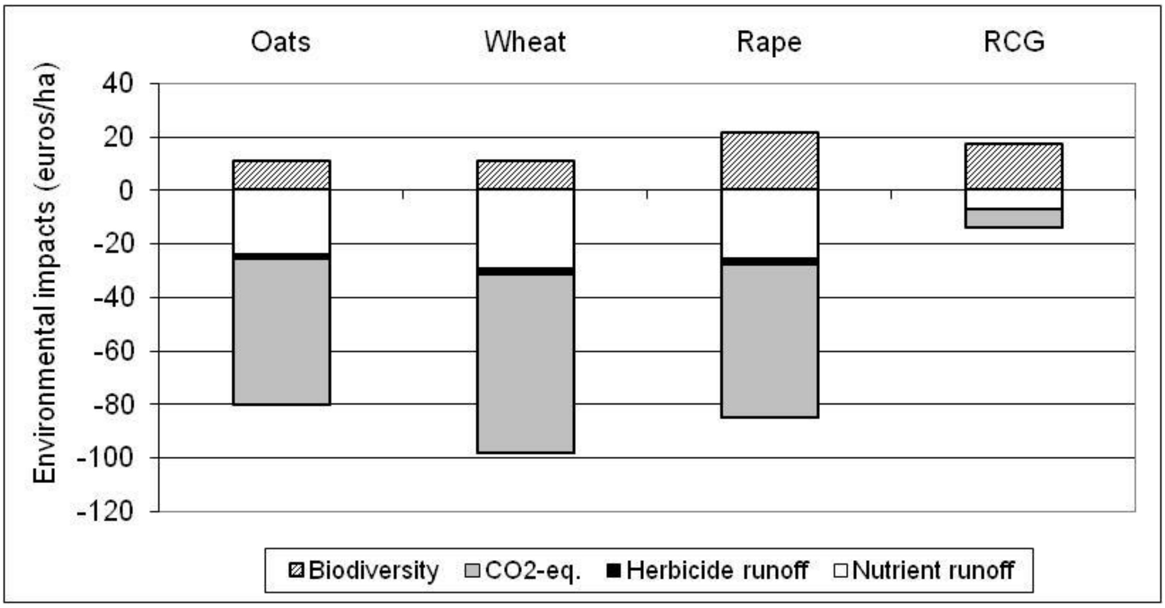

Figure 1. *Environmental profile of alternative land uses in the Baseline scenario, €/ha.*

Concerning the environmental effects, figure 1 illustrates that reed canary grass (RCG) performs well. Its good environmental performance is mainly driven by its low CO_2-eq emissions. This is largely explained by the fact that RCG is a perennial crop that sequesters carbon and thus soil CO_2 emissions are in fact negative, whereas for other crops, which are annual crops and cultivated with conventional tillage, soil CO_2 emissions are significant. Moreover, RCG is cultivated with low fertilizer intensity and thus low CO_2-eq emissions related to fertilizer use. Because of high fertilizer and herbicide use intensity wheat performs poorly with respect to both CO_2-eq emissions and nutrient runoff. With respect to the biodiversity benefits provided, rape is the highest ranked of the land use types in the Baseline scenario. This is because the wildlife habitat index uses butterflies as the key species and as such rape provides a higher quality habitat than cereals. The overall environmental performance of alternative land use types is mainly driven by the value of CO_2-eq emissions and nutrient runoff damage. Herbicide use intensity and resulting herbicide runoff damage have only a marginal effect on the environmental performance of alternative land use types. Incorporation of biodiversity benefits favour rape and reed canary grass over cereals.

Concerning social welfare (defined as the combination of environmental effects and farmers' private profits), figure 2 illustrates the social profitability of alternative land uses without government support in the Baseline. The results show that the land use type that delivers the best environmental performance (reed canary grass) is the least profitable for farmers. Overall, first generation biofuel rape for biodiesel provides the highest ex-post social welfare, since it provides a combination of the highest farm profits with the second lowest negative net environmental impact. This social welfare ranking illustrates that ex-post social welfare of alternative land use types is mainly driven by profitability of land use rather than the social valuation of environmental effects.

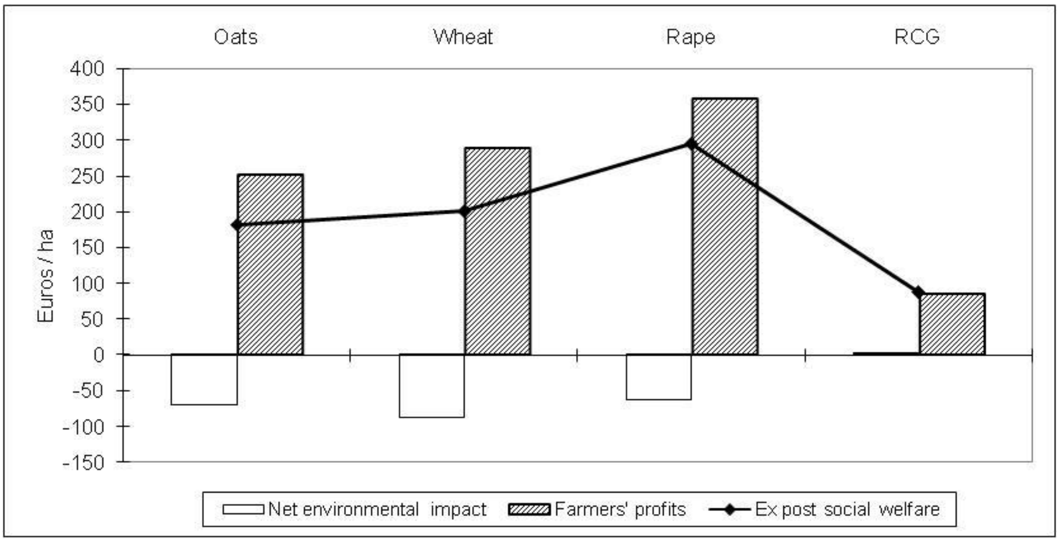

Figure 2. *Social welfare under alternative land uses in the Baseline scenario, €/ha.*

Extending the analysis to the ex-post social welfare estimates for alternative policy scenarios, the results are presented in Figure 3.

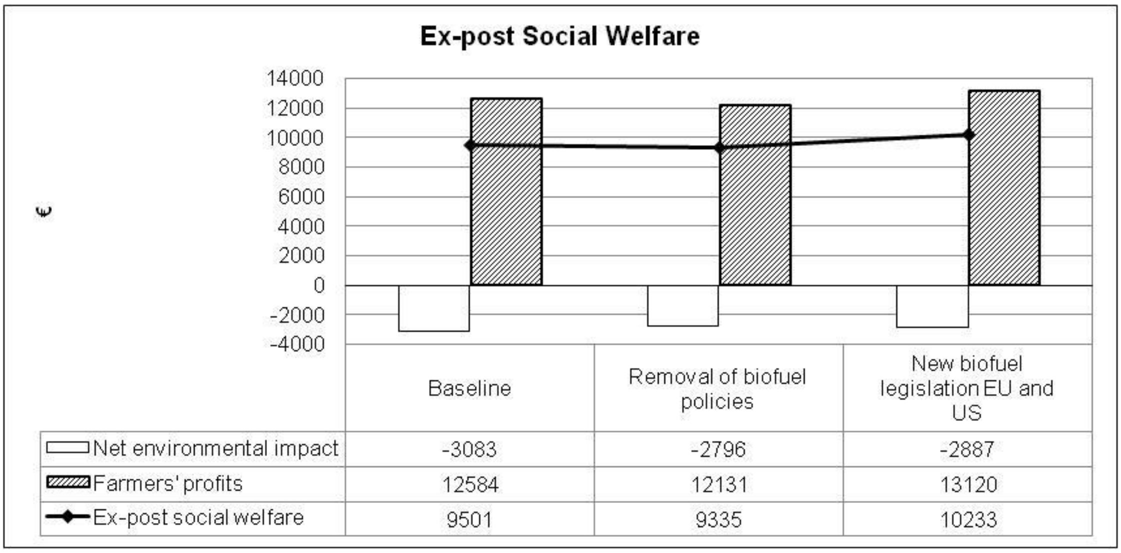

Figure 3. *Ex-post social welfare under alternative scenarios, €.*

Figure 3 shows that the ex-post social welfare of alternative policy scenarios is driven mainly by profitability rather than the social valuation of environmental effects.

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APPENDIX

Figure A1. Environmental effects covered in the empirical application.

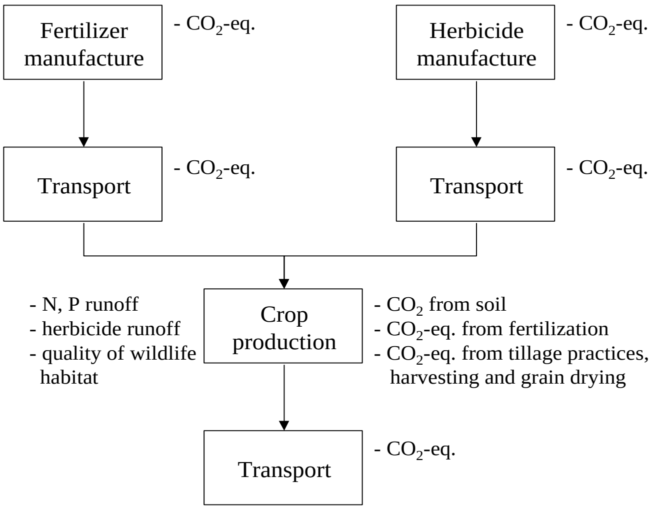

Table A1. Agricultural Price, Cost, and Support Parameters

Parameter	*Symbol*	*Value*
Effective output price (net of grain drying and transportation costs) in Baseline	*p*	
Spring wheat		€ 0.0905/kg
Barley		€ 0.0825/kg
Oats		€ 0.0725/kg
Rape		€ 0.2020/kg
Reed canary grass		€ 0.0152/kg
Price of nitrogen fertilizer	*c*	€ 1.30/kg
Price of herbicide	α	€ 6.7/kg
Labour cost (barley, spring wheat, oats and rape) Reed canary grass	*wh*	€ 155/ha € 39/ha
Fixed cost of machinery	*K*	€ 242/ha
Expenditure for other inputs than fertilizer and herbicide	Θ	
Spring wheat		€ 129/ha
Barley		€ 90/ha
Oats		€ 60/ha
Rape		€ 117/ha
Reed canary grass		€ 125/ha
Fuel costs (barley, spring wheat, oats and rape)		€ 54/ha
Fuel costs (reed canary grass)		€ 5/ha
Support payments	*S*	
SFP		€ 230/ha
LFA		€ 169/ha
Agri-environment		€ 117/ha
Energy crop payment		€ 45/ha

Table A2. **Baseline, Policy scenario 1, and Policy scenario 2**: *land allocation, input use intensity, production and farmers' profits*.

Crop	Land area, ha	Nitrogen use, kg/ha	Herbicide use, kg/ha	Production, kg/ha	Total production, kg	Profits, €/ha	Total profits, €
	Baseline						
RCG	2	33.7	-	4 609	9 219	221	443
Oats	4	72.4	0.82	3 112	12 449	226	903
Wheat	21	130.2	0.91	3 397	71 327	263	5 513
Rape	15	93.8	0.96	1 749	26 229	333	4 997
Total	42	-	-	-	119 224	-	11 856
	Policy scenario 1 – Removal of biofuel support						
Oats	24	74.2	0.84	3 272	78 535	237	5 698
Rape	18	88.6	0.94	1 700	30 602	299	5 382
Total	42	-	-	-	109 137	-	11 080
	Policy scenario 2 – New biofuel legislation EU and US						
RCG	4	39.6	-	4 913	19 651	236	944
Wheat	18	131.1	0.92	3 316	59 688	267	4 802
Rape	20	94.3	0.97	1 706	34 120	348	6 968
Total	42	-	-	-	113 459	-	12 714