PARALLEL SESSIONS 4

4C / Specific impact categories of the primary sector

Combining Life Cycle Assessment and Linear Programming to explore sustainable farming regions

Ivonne Acosta-Alba[1,2,*], Santiago Lopéz-Ridaura[3], Hayo M.G. van der Werf [1,2]

[1]INRA, UMR1069, Soil Agro and hydroSystem, F-35000 Rennes, France
[2]Agrocampus Ouest, UMR1069, Soil Agro and hydroSystem, F-35000 Rennes, France
[3]INRA, UMR Innovation, UMR951, F34060, Montpellier, France

ABSTRACT

Life Cycle Assessment (LCA) is a helpful tool in the development of low-impact agricultural systems at the farm scale. This tool, however, is rarely used at the scale of a farming region. We included LCA-based environmental impact indicators into a Multi Goal Linear Programming (MGLP) model to identify and describe trade-offs between LCA indicators and other indicators of sustainability. Understanding these trade-offs can trigger and support decision making (López-Ridaura *et al.*, 2005). We used this concept to propose different scenarios of an optimized and environmentally sustainable dairy region in Brittany, quantifying trade-offs between scenarios and indicators.

Keywords: Sustainability, Life Cycle Assessment (LCA), Multi Goal Linear Programming (MGLP), agricultural production, territory.

1. Introduction

Life Cycle Assessment (LCA) is a helpful tool in the development of low-impact agricultural systems. It is often used to assess whether a system pollutes more than another (e.g. conventional vs organic) or to compare technologies to reduce environmental burdens.

Assessing sustainability implies not only identifying systems with reduced environmental burdens but also considering socio economic indicators and contextualising systems at different scales (López-Ridaura *et al.*, 2005) Sustainability assessment involves the definition of functions and goals. Today, agricultural systems should not only produce nutritional energy, proteins, commodities and income for farmers, but they should also respect several environmental regulations and pollution reduction goals (such as reducing nitrate leaching). These goals can be expressed by objectives to be attained. Considering these goals as reference values representing a way towards sustainability might help us to guide current agricultural systems towards sustainability (Acosta-Alba *et al.*, Submitted).

However, which method should be used to explore the consequences of the implementation of these objectives for the production of a farming region[1]? How to evaluate sustainability at different spatial scales using LCA results? The consideration of results from assessments at the farm scale and for environmental indicators only, may lead to simplistic solutions. It would be useful to find a way of including other indicators, at a larger scale.

We suggest to use LCA indicators in a Multiple Goal Linear Programming (MGLP) model including other socio economic indicators to produce scenarios of evolution for the farming region and identify trade-offs between indicators. This articulation of approaches might allow a wider vision about the development of agricultural areas and, by means of scenario analysis, support discussion between stakeholders.

* Corresponding Author. e-mail: ivonne.acosta-alba@rennes.inra.fr

[1] A farming region is an identified geographic entity, differentiated and structured by the activities and social groups which occupy it and interact there (Papy, 2001). In France the term used is “agricultural territory”.

2. Materials and methods

MGLP has been used in many applications, such as forest management (Yousefpour *et al.*, 2010) agricultural and natural resources planning (De Wit *et al.*, 1988; van Ittersum *et al.*, 1998; van Calker *et al.*, 2007) and industrial and transportation problems (Alvarez *et al.*, 2010). It has also been used to integrate information at different scales and quantify trade-offs between indicators (López-Ridaura *et al.*, 2005; Meyer *et al.*, 2009). Van Ittersum *et al.*, (1998) describe scenarios as an "approach to investigate combinations of exogenous conditions, preferences for objectives and technical feasibilities". MGLP can allocate limited resources between several alternatives of land use to generate land use scenarios.

The basic structure of the MGLP model used (Lopéz-Ridaura *et al.*, 2005; van Ittersum *et al.*, 1998) has the form of a standard linear programming model. The model applied in this study was written as a linear programming model and solved with the General Algebraic Modelling System (GAMS), a modelling system for mathematical programming and optimization (Brooke *et al.*, 1992; Rosenthal, 2006). The optimization models simulate scenarios satisfying more that one goal at the same time within the constraints set.

We assessed 40 dairy farms in Brittany (western France), a region where agriculture represents around 60% of total land use and has major environmental impacts (Merot *et al.*, 2009). Farms were assessed with EDEN-E, a LCA-based tool for dairy systems (van der Werf *et al.*, 2009). The implementation of EDEN-E requires a survey to assess farm inputs and outputs, the housing and manure management systems, techniques used for manure application, crops grown, and grazing and feeding strategies.

In the LCA we focus on the major environmental impacts of dairy production systems in Brittany: (i) Climate change, a global impact, is represented by Greenhouse Gas emissions(GHG) in terms of CO_2-equivalents ha^{-1}; (ii) Water quality, a local impact, is represented by the nitrate leached (NO_3) in terms of kg of N-NO_3 ha^{-1}, and (iii) Non renewable energy use (EU), a resource depletion impact, is quantified in terms GJ ha^{-1}. These indicators describe the environmental efficiency of farms and they are all expressed per hectare of land occupied.

Besides the environmental efficiency, we selected four other criteria describing the sustainability of dairy farms: autonomy, economic viability, social contribution and productivity. For each criterion, several indicators were chosen (Table 1). Each of these indicators represents an objective for minimization or maximization, or a constraint in the MGLP model. For productivity, milk, and other products were taken into account. Products were expressed as nutritional energy from crop and animal products.

Farms were grouped in types. The main criterion for grouping them was the production method, farms being specialized or not. The degree of farm specialization is assessed by the fraction of income from an activity other than meat or milk. Non specialized farms sell crops (Table 2). According to their degree of intensification, farms were grouped in 7 types ranging from intensive to organic. These farm types make up the current configuration of the region (around 2700 hectares). The total area of the 40 farms was used as a virtual region composed exclusively of farms.

Table 1: Criteria and indicators used to represent dairy farms. *Objective: arrows indicate whether the indicator is minimized ⬇ or maximized ⬆.

Criterion	Indicator	Dimension	Objective*
Autonomy	Additional Area, i.e. off-farm land for feed (AA)	Ha	⬇
	Total nitrogen inputs (N)	kg of N ha^{-1}	⬇
Environmental efficiency	Nitrate leached (NO_3)	$KgN\text{-}NO_3ha^{-1}$	⬇
	Non-renewable energy use (EU)	GJ ha^{-1}	⬇
	Greenhouse Gas emissions (GHG)	CO2 teq ha^{-1}	⬇
Economic viability	Production cost (COST)	KEuros ha^{-1}	⬇
	Total income (Income)	KEuros ha^{-1}	⬆
	Gross margin (GM)	KEuros ha^{-1}	⬆
	Gross Operating Surplus (GOS)	KEuros ha^{-1}	⬆
	Farms profit (FP)	KEuros ha^{-1}	⬆
Social contribution	Employment (Employ)	workers ha^{-1}	⬆
Productivity	Milk produced (MILK)	t ha^{-1}	⬆
	Energy efficiency (EE)	no unit	⬆
	Nutritional energy, animal products (NE_AP)	GJ ha^{-1}	⬆
	Nutritional energy, crops products (NE_CP)	GJ ha^{-1}	⬆
	Total nutritional energy (NE)	GJ ha^{-1}	⬆

Table 2: Characteristics used to define types of dairy farms.

Production	Income from dairy household. (%)	Other products sold	Dairy system intensification		Type	N° of farms
			Milk (t/ha)	Concentrate feed (g/kg milk)		
Organic	100 %	-	-	-	1.Organic (Org)	5
Conventional	≥ 90 %	-	> 7	-	2.Specialized Intensive (Sint)	5
			≤ 7	≥ 100	3.Specialized Semi-intensive(Ssi)	2
				< 100	4.Specialized extensive (Sext)	3
	< 90%	Crops	> 7	-	5.Dairy-Crops Intensive (DCint)	12
			≤ 7	≥ 100	6.Dairy-Crops Semi-intensive(DCsi)	6
				< 100	7.Dairy-Crops Extensive (DCext)	7

The current configuration of the region is characterized by the existing proportions of each of the 7 farm types. We explored other configurations, corresponding to more sustainable agricultural scenarios. A sustainable scenario implies the accomplishment of predefined objectives. The current situation was compared with four main scenarios implying the (i) the maximization of milk production with no constraints ("max milk") (ii) the maximization of milk production setting a maximum of level of 50 mg l^{-1} for the NO3 indicator ("50 mg/l") (iii) the maximization of milk production setting a maximum of level of 25 mg l^{-1} for the NO3 indicator ("25 mg/l") and (iv) the maximization of nutritional energy production with no constraints ("Max NE"). Finally, trade-off curves between conflicting indicators were drawn with the MGLP model by gradually relaxing the constraint in the value of one indicator while maximizing or minimizing another. The current performance of the region was placed below this curve to identify the window of opportunities to strengthen the sustainability of the region.

3. Results

Figure 1a represents the results of the scenarios in terms of the normalized value of indicators. The closer the indicator value is to 100, the better the value corresponds to the objective (minimization or maximization). Figure 1b shows the resulting land use for each of the scenarios. The scenario 25mg/l has the best performance in terms of autonomy and environmental impact. However, for indicators related to productivity and economy it is the worst scenario (except for the cost indicator). This scenario was not feasible with the current farms unless production is abandoned on 1.5% of the region (figure 1b). The "Max Milk" scenario is the best in terms of income and gross margin and it would imply to have exclusively "Specialized Intensive" farms. The 50 mg/l and max Milk scenarios have a similar profile, although the proportions of farms are different. The Max NE scenario has the best performance in terms of energy production due to nutritional energy from crop products.

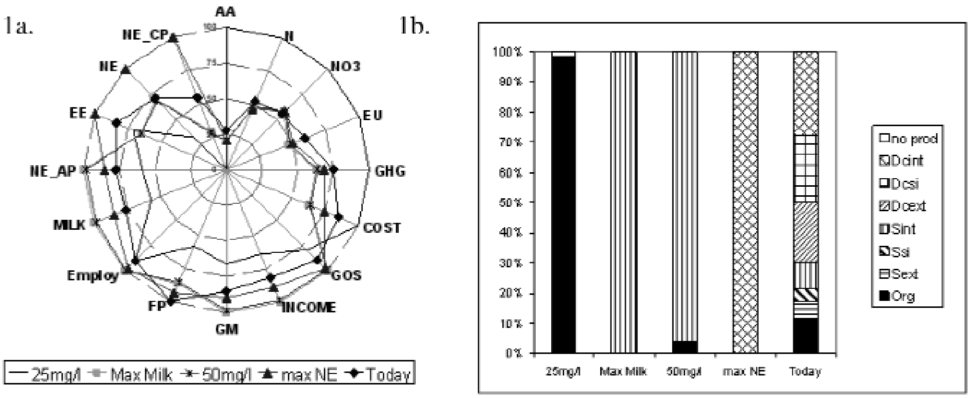

Figure 1: 1a. Representation of scenarios in a radar graph. 1b. Land occupation by farms' types.

MGLP can also be useful to quantify trade-offs between indicators. Figure 2 shows the trade-off between Nutritional Energy production (NE) (being maximized) and Greenhouse Gas Emissions (GHG) (set as a constraint at different levels). Maximum NE (100 425 GJ) is associated with at least 20 544 tones of GHG. When GHG is set as a very strong constraint (below 14 500 tones), the maximum NE is 33 800 GJ, at one third of its maximum. The current situation is situated below this curve, revealing that there is room for improvement in the performance of the region: (i) production can be increased at the same level of GHG (vertical arrow) and (ii) at the same level of production; GHG can be considerably decreased (horizontal arrow).

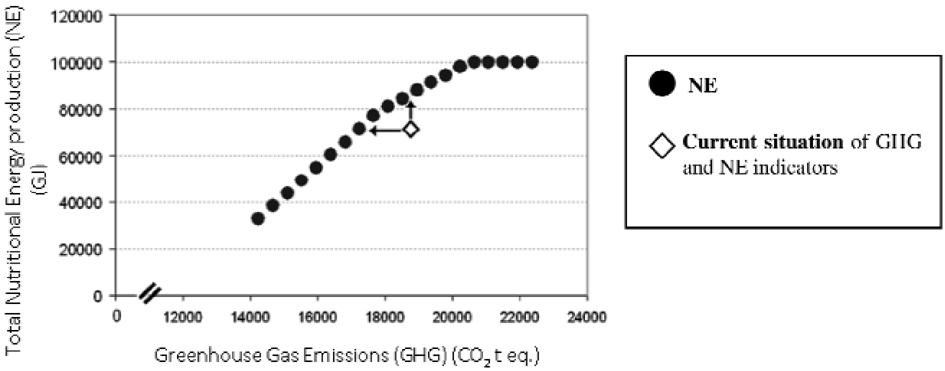

Figure 2: Trade off curve between nutritional energy production (NE) (indicators maximised) and greenhouse gas emissions (GHG) (set as constraint).

The area between the arrows represents the window of opportunities for increased performance.

4. Discussion

The presented attempt to use LCA results in a sustainability assessment is above all a methodological and explorative study. We worked with farm types, since small changes in the farming system (feed used, crop rotation) can perturb the coherence of the system and have huge impacts on production, natural resource consumption and emissions of pollutants. In a "real" region it is possible to convert an organic farm into a conventional one or the opposite but this might have variable socio-economic consequences. Therefore MGLP seems a good tool to explore the range of possible optimization options of agricultural systems.

Using indicators for five complementary criteria describing sustainability is useful identify trade-offs and may facilitate the comprehension and the discussion with stakeholders. The window of opportunities described by the trade off curve and the current situation is a starting point to quantify to what extent we may expect to improve the current situation with real farms. It is useful also to measure the socio-economic consequences of environmentally friendly systems.

MGLP is easy to adapt and to use to explore the best performance of current and alternative farming systems and their efficiency. A profound knowledge of the region and its history is, however, very important to assess possible evolutions and current trends. For instance, the scenarios "Max milk" and "50mg/l" were close, probably due to increasingly strict environmental regulations in this region (Merot et al., 2009). Moreover, today scenario has the best farm profit (FP) indicator. This strengthens the premise of a farming region which has faced several constraints, improving its global efficiency over the time. Our study illustrates, however, that stricter environmental regulations (25mg/l) may be difficult to implement in regions where agriculture is the dominant land use.

We are at present examining the effects of introducing additional goals (economic and social objectives), new land use activities (e.g. forest to capture C) and including different consumption patterns to adequate agricultural production to healthy diets.

From an LCA methodological point of view, using LCA indicators in this study implied expressing these indicators per hectare. To avoid exporting impacts elsewhere, we have used an indicator called "Additional area" or "Off-farm area" which assesses the area needed to produce feed imported on the farm. The use of MILK and NE indicators reveals the trade-offs between products. Besides, NE considers all farm products and responds to the need to connect agricultural and nutritional sciences (Welch and Graham, 1999; Kratochvil *et al.*, 2004; Peters *et al.*, 2007).

5. Conclusion

The major benefits of using MGLP coupled to LCA are: (i) the generation of scenarios that can be discussed with stakeholders as support information for environmentally-conscious decision making (ii) better understanding of sustainability to optimize agricultural production from an LCA perspective (iii) associate different complementary production modes in a region (not only organic or intensive farms) (iv) taking into account social and economic indicators while considering environmental concerns.

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Allocating greenhouse gas emissions from land conversion

Tommie Ponsioen[1,*], Hans Blonk[1]

[1]Blonk Milieu Advies (Blonk Environmental Consultants), Gouda, the Netherlands,

ABSTRACT

Deforestation causes large amounts of greenhouse gas emissions, but there is no adequate method yet for allocating those emissions to the different economic activities that follow deforestation. We present a method that divides emissions from annual deforestation rates for agricultural expansion in a country between timber harvest and agricultural activities, based on possible income from selling timber and agricultural land use returns. The emissions that are allocated to agricultural activities are only divided between those activities that show a trend of area expansion, in proportion to the sum of those expansion trends. Although this method is not perfect, it works with publicly available data, it can be adapted when more detailed information is available, and it gives more realistic and fairer results than the currently used method(s). It could also provide better grounds for motivating producers and consumers to improve their behaviour in relation to greenhouse gas emissions.

Keywords: Greenhouse gas emissions; Land conversion; Agricultural activities; Timber; Attributional life cycle assessment,

1. General information

Deforestation causes large amounts of greenhouse gas emissions, because of burning and decay of natural biomass and forest products (fuel wood). There can be several economic purposes for deforestation. Initially, quality timber is selected for wood production and other biomass is sold as fuel wood or for charcoal production. Then the land is either left fallow or is used for agricultural land use activities, such as crop growing (monoculture or crop rotations) and cattle grazing. If left fallow, secondary forest can develop or the land is eventually used for agricultural activities (Figure 1).

The Amazon rainforest and the Cerrado forest in Brazil are two of the most important biomes that have been subject to large-scale deforestation in the past fifty years (Margulis, 2004). Many scientists believe that potential income from livestock grazing and soybean cropping is the most important incentive for deforestation in Brazil. According to life cycle assessment methodology, the greenhouse gas emissions from deforestation should be allocated to all activities that create economic value and that are related to the deforestation. However, several problems prevent us from doing this correctly. First, data and information is not always available because of large-scale illegal practices or company confidentiality. Second, any time period after deforestation within which agricultural activities are still considered to be related to deforestation is arbitrary.

The most widely used method for allocating greenhouse gas emissions from deforestation, referred to as direct land use change (PAS2050, 2008; European Parliament, 2009), ignores income from timber/fuel wood and dictates an amortization period of twenty years during which the emissions are allocated in equal measure to those years, regardless of whether ag-

* Corresponding Author. e-mail: tommie@blonkmilieuadvies.nl

ricultural activities take place or not. This means that even if no more land conversion takes place, it takes years before an agricultural product is free of attributed emissions from deforestation. Moreover, detailed information is needed, which is difficult to obtain, such as the exact location where the agricultural product under study was produced and when the land was converted. When such information is lacking, for example when only the country where the crop was grown is know, worst case situations are applied. This results in three problems: (1) large differences occur between products that belong to the category of within the amortisation period and products for which can be proven they do not, (2) the sum of the carbon footprints (attributed greenhouse gas emissions) of all land use activities is not in line with calculations on country or global scales and (3) the method does not take any actual displacement effects into account. The lack of an adequate method for allocating greenhouse gas emissions from deforestation to the different economic activities that follow deforestation motivated us to develop a new method.

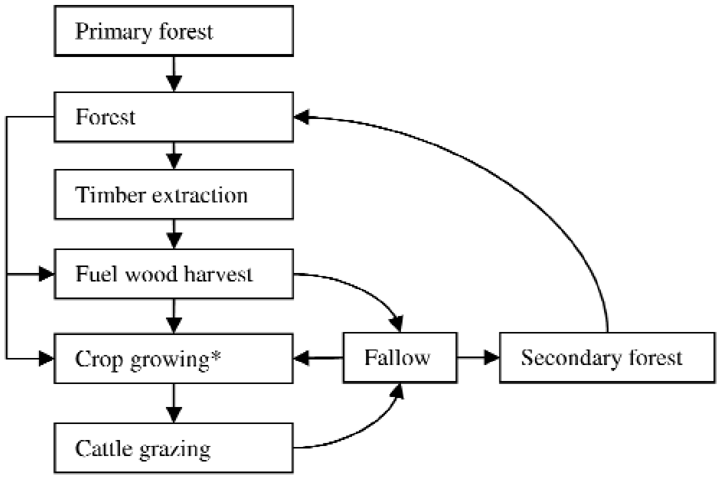

Figure 1: Schematic overview of activities that can be related to deforestation and forest ecosystems (* crop growing can be monoculture or crop rotations)

2. Methods

Our method considers annual deforestation rates for agricultural expansion in a country. Biomass estimates of forests can be used to calculate the total greenhouse gas emissions per hectare. According to the IPCC (Paustian *et al.*, 2007), about 1.8 kg CO_2 equivalents per kg biomass are emitted from burning forest. This is equal to the amount of CO_2 that would be released from the oxidation of carbon in biomass. We used this factor for biomass decay as well, assuming that methane emission from biomass decay does not significantly contribute to the total emissions.

The calculated annual greenhouse gas emissions from deforestation in a country are divided between different economic activities. First, the emissions are divided between timber harvest and agricultural land use activities, based on the economic value of timber and of cleared land for agricultural purposes. For timber, prices can be used. Prices of cleared land, on the other hand, do not necessarily represent the economic value, because of a lacking or underdeveloped land market (no clear land ownership and few documented transactions). Therefore, we suggest the use of agricultural returns converted to net present value. The emissions that are allocated to agricultural land use activities are divided only between those activities that increased in area. However, because the use of actual annual increases could result in very high fluctuations in the allocation fractions from year to year, we propose the use of expected increases from a trend analysis. The allocation fractions are then equal to the area expansion trends in proportion to the sum of those expansion trends. We applied this method to deforestation in Brazil and assumed that the agricultural area expansion rate in Brazil is equal to the deforestation rate for agricultural expansion.

3. Results

The average natural biomass in Brazilian forests is about 280 tonnes per ha: about 75% tropical forest containing 300 tonnes of biomass per ha, and 25% other forest containing about 220 tonnes per ha (FAO 2001; Paustian *et al.* 2007). The greenhouse gas emissions from deforestation are therefore about 500 tonnes CO_2 equivalents per ha: 280 tonnes per ha multiplied by 1.8 kg CO_2eq per kg biomass (Paustian *et al.* 2007).

The average volume of timber that is extracted from deforestation areas in Brazil is about 20 m^3 per hectare and its stumpage value is about US$ 13 per m^3. The average income from timber extraction is therefore about US$ 250 per deforested hectare. An analysis by Grieg-Gran (2008) gives a realistic indication of the value of cleared land based on agricultural land use returns from deforested areas (converted to net present value in the year 2007 with a discount rate of 10% and a time horizon of 30 years), which amounts to about US$ 460 per hectare (Table 1). The allocation fraction for timber is therefore calculated to be 0.35 (250 US$/ha/[250 US$/ha+460 US$/ha]), and so the allocation fraction for agricultural land use activities is 0.65.

Table 1: Deforested land use returns in Brazil (source: Grieg-Gran 2008)

Land use	**Returns (US$/ha)**	**Area (1000 ha)**	**Returns (million US$)**
Beef cattle, medium/large scale	413	1955	807
Beef cattle, small scale	3	217	1
Dairy	172	217	37
Soybeans	3278	155	508
Manioc/rice	3	496	1
Perennials	3	31	0
Tree plantations	2550	31	79
Total agriculture	462	3102	1433
One-time timber harvesting	251	3102	779

Using FAO (2010) statistics, the trend analysis of area expansion resulted in 0.64 million ha per year for soybean, 0.14 for sugar cane and 0.12 for other crops with expanding area between 1987 and 2008. This period was chosen as most representative for the expected trend (Figure 2). The area of meadows and pastures also expanded during that period, but as there is a clear trend towards stabilisation no expansion is expected in the meadow and pasture area in Brazil. This results in a relative area expansion for soybean of 0.71 (0.64/[0.64+0.14+0.12]). The total expected agricultural area expansion is 0.54 million ha per year. This means that part of the area expansion for soybean, sugarcane and other crops with expected expanding area is due to the contraction of the area under other crops, such as rice, beans, cotton, wheat and coffee. This part is equal to 0.36 million ha per year divided by 0.90 million ha per year, which yields 0.40.

The greenhouse gas emissions that can be allocated to a hectare of soybean is therefore 0.65 allocated to agricultural land use activities, multiplied by 0.6 [= 1 - 0.40] from deforestation, multiplied by 0.64 million ha per year, and divided by the actual soybean area (22 million ha in 2010 according to the trend), which gives 5.7 tonnes CO_2 equivalents per ha (Table 2 shows the parameter values and stepwise calculations). For comparison, the greenhouse gas emissions from agricultural input production and emissions during crop growing are about 1.4 tonnes per ha (according to own calculations).

Table 2: Parameter values and stepwise calculations of the land conversion carbon footprint of soybean production in Brazil

Parameter	Value	Units
Emissions from deforestation (a)	500	tonnes CO_2eq/ha/year
Allocation fraction to agriculture (b)	0.65	-
Fraction expansion from forest (c)	0.6	-
Expected soybean expansion (d)	0.64	10^6 ha/year
Soybean area in 2010 (e)	22	10^6 ha
Land conversion carbon footprint (a × b × c × d/e)	5.7	tonnes CO_2eq/ha

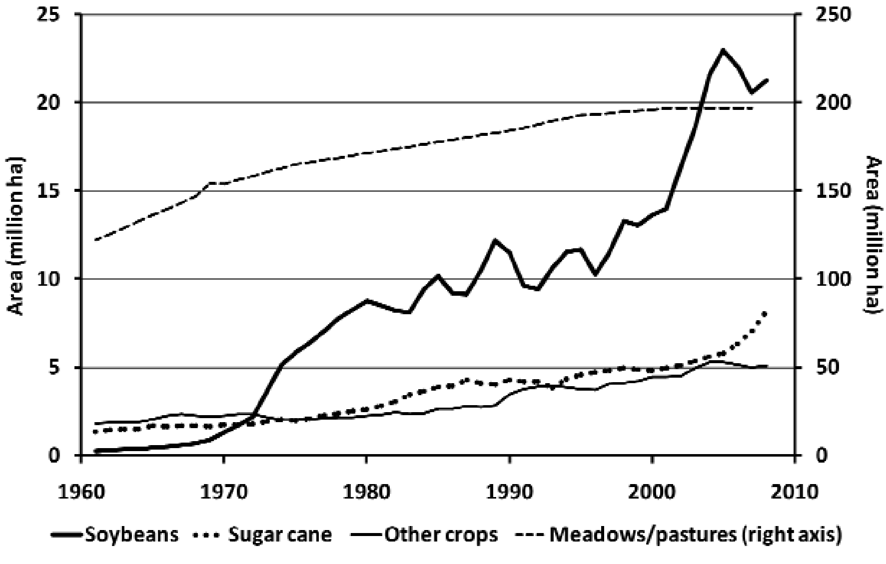

Figure 2: Area of soybeans, sugar cane and other crops with expanding area and meadows/pastures between 1961 and 2008 (source: FAO 2010)

4. Discussion

The new method presented in this paper was designed to allocate actual greenhouse gas emissions from deforestation for agricultural expansion to different economic activities. One problem with the method that could not be solved is that it may underestimate the deforestation rates for agricultural expansion (assuming expansion rates are equal to deforestation rates), because after a period of agricultural activities land may be left fallow (Figure 3). On the other hand, if secondary forest develops and is left unharmed, carbon dioxide is sequestered and the emissions are partly allocated to timber/fuel wood. If the land is used again for agricultural activities, the earlier underestimation is compensated because that type of expansion is then considered as deforestation for agricultural expansion.

The data requirements for the presented method is less intensive than other methods that focus on land displacement effects (e.g. Searchinger *et al.*, 2008). Moreover, those methods are based on marginal analysis, which is prone to debatable assumptions. The existing method that divides the emissions from deforestation over the first twenty years after the event, requires data on the exact location of agricultural production and the history of the used land. Most of the required data for the presented method is publically available from FAO publications (e.g. FAO 2001) and statistics website (faostat.fao.org) and from the IPCC Guidelines (e.g. Paustian *et al.* 2007). Data for allocation between round-wood and agriculture is more difficult to obtain. The study by Grieg-Gran (2008) provides data for the most important countries, where large scale land conversion takes place (for example Brazil, Indonesia and Malaysia). For other countries, additional analyses may be required.

Despite these imperfections, we believe that the method gives more realistic and fairer results than other methods. The results are more realistic because the method only considers annual emissions rather than fractions of emissions that occurred within a twenty year period in the past. The results are fairer because deforestation is profitable for both timber exploitation and agriculture, and the greenhouse gas emissions that are allocated to agricultural are divided between the land use activities that cause most pressure on land.

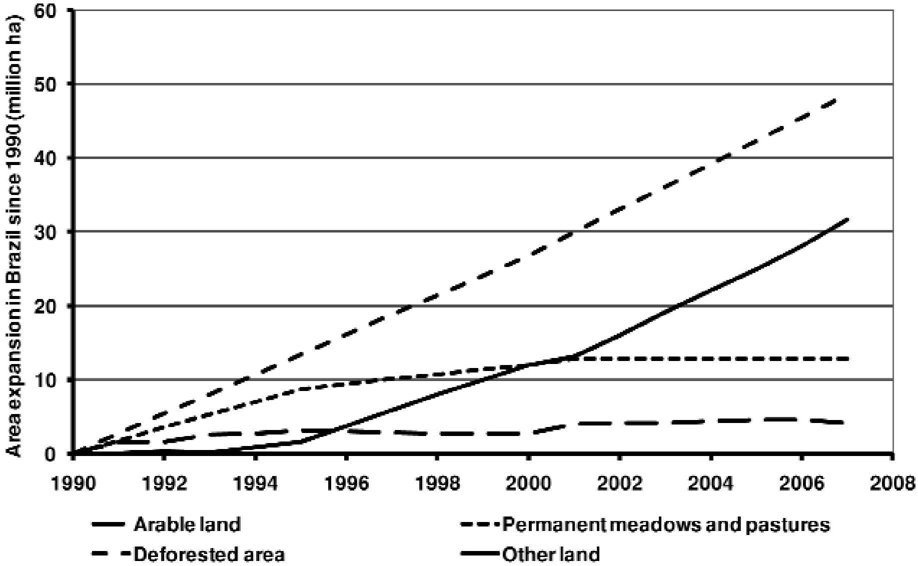

Figure 3: Area expansion in Brazil since 1990 (source: FAO 2010; other land and arable land and permanent crops corrected between 1990 and 1994; other land includes land that has been left fallow for more than five years)

5. Conclusion

We developed a method for allocating greenhouse gas emissions from land conversion to agricultural land use activities. Although this method is not perfect, it works with publicly available data, it can be adapted when more detailed information is available, and it gives more realistic and fairer results than the currently used method. It could also provide better grounds for motivating producers and consumers to improve their behaviour in relation to greenhouse gas emissions.

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Developing new methodology to assess direct and indirect impacts of agricultural activities on soil quality

Emmanuelle Garrigues[1,2,*], Michael S. Corson[1,2], Christian Walter[1,2], Denis A. Angers[3], Hayo van der Werf[1,2]

[1] INRA, UMR1069, Soil Agro and hydroSystems, F-35000 Rennes, France
[2] Agrocampus Ouest, UMR1069, Soil Agro and hydroSystems, F-35000 Rennes, France
[3] Agriculture and Agri-Food Canada, 2560 Blvd. Hochelaga, Quebec City, Quebec, G1V 2J3, Canada

ABSTRACT

The aim of this study is to propose a new impact indicator of agricultural practices on soil quality in LCA. Currently, few LCA methods consider the impacts of agricultural activities on soil quality. Consensus has not yet been reached on determining a Minimum Data Set (MDS) of soil characteristics to define soil quality and on combining them into a synthetic indicator because of the numerous interrelated soil properties and their complex link with agricultural practices. We propose a MDS of soil characteristics, as influenced by climate and agricultural management characteristics, to assess impacts of maize production on soil quality in two locations in France and Brazil. The method presented here is flexible enough to assess different types of soil and climate. Soil erosion and effects on organic matter are the first aspects of soil quality considered. Off-site effects on soil quality will be taken into account in future development of the method.

Keywords: LCA, soil quality, Minimum Data Set, land use, agricultural assessment.

1. Introduction

Soil quality is an important element of ecosystem sustainability and agricultural production. It is a dynamic phenomenon that refers to inherent properties, human use and management of soil, and has to be assessed to measure its changes. Several soil-quality definitions have been proposed. The Soil Science Society of America defines it as "the capacity (of soil) to function" (Karlen *et al.*, 1997). The most important functions cited by Andrews *et al.* (2004) are water flow and retention, solute transport and retention, physical stability and support, retention and cycling of nutrients, buffering and filtering of potentially toxic materials, and maintenance of biodiversity and habitat. Soil properties can be associated with these functions to define soil quality, but soil characteristics are numerous and interrelated and have a complex link with agricultural practices. Contemporary soil-quality assessment often focuses on determining a Minimum Data Set (MDS) of soil characteristics to select the most appropriate components describing physical, chemical and biological properties. Until now, no synthetic indicators have been proposed that would combine the main soil properties into a simple formula valid for all types of soils and climates. In the MDS proposed in the literature, soil organic matter, texture and density are almost unanimous (Kelting *et al.*, 1999; Dexter, 2004; Wienhold *et al.*, 2004; Masto *et al.*, 2008) among numerous other physical and chemical properties. The biological properties of soil can be taken into account directly (Arshad and Martin, 2002; Bohanec *et al.*, 2007; Kaschuk *et al.*, 2010) or indirectly by assuming a correlation between the size of soil microflora and the content of organic matter in mineral soils (Kirchmann and Andersson, 2001).

[*] Corresponding author: Emmanuelle.Garrigues@rennes.inra.fr

The question of the environmental impact of agriculture is treated through various methods of assessment at the farm scale (van der Werf and Petit, 2002). Some aspects of soil quality are considered in few of them. The scoring method INDIGO proposes the assessment of potential impacts of arable farming systems on the environment through different agro-ecological indicators, such as organic matter (Girardin *et al.*, 2000). The indicators have to be studied together since they provide a "control panel" for the farm. The method called IDEA (French acronym for "Farm Sustainability Indicators") (Zahm *et al.*, 2008) assigns scores to farmer production practices and farmer behaviour and is conceived as a self-assessment grid. The indicators must be adapted to local farming practices before using the IDEA method (Zahm *et al.*, 2008). Erosion and heavy metal emissions to the soil are considered. None of these methods, however, takes into account upstream processes impacts at regional or global scales; in contrast, Life Cycle Assessment (LCA) does. Furthermore, the LCA method is flexible enough to capture several soil properties and management practices in various combinations to assess impacts on soil quality.

Yet, analysing impacts on soil quality remains one of the unresolved problems in LCA because of its spatial and temporal variation and local environmental uniqueness (Reap *et al.*, 2008). It is one issue listed as important in the assessment of the impacts of fertile land use in "Life Support Functions" of LCA (Lindeijer *et al.*, 1997). Milà i Canals *et al.* (2007) developed an approach suggesting soil organic matter as the sole indicator of soil quality, which could be applied in agro-forestry, when soil quality is not compromised by other impacts such as acidification, salinisation, etc. The method called SALCA-SQ (Swiss Agricultural Life Cycle Assessment for Soil Quality) developed for Swiss conditions, considers physical, biological and chemical characteristics of the soil with nine indicators (Oberholzer *et al.*, 2006). This method is focused on direct (on-farm) impacts, and thus does not consider indirect impacts on soil quality that may occur elsewhere due to the production of farm inputs, such as concentrated feed.

As identified by Milà i Canals *et al.* (2006) in the assessment of land use impacts in LCA, the aim of our study is to use LCA to aid agricultural management decisions. We focus on the effects of management practices on soil quality through new mid-point impact categories in LCIA profiles. In this paper we describe the preliminary steps of a conceptual and operational approach applied to maize production in France and Brazil.

2. Methods

Our approach to assess the impacts of a given production system's agricultural activities on soil quality is progressive and iterative. It is based on: (1) development of a decision tree of the choice of the midpoint impact categories contributing to the assessment of soil quality according to the pedoclimate conditions; (2) definition of algorithms to assess impacts on soil, including definition of the MDS and the management data used; and (3) aggregation of characteristics deducted into indicators of soil quality. The definition of the MDS has to be defined before development of a decision tree because it incorporates this MDS. So, the second step is treated first and described in this paper.

In the choice of the soil MDS, the appropriate indicators have to meet the management goals, but not only: inherent properties not influenced by management are also considered because they are involved in the definition of the soil sensitivity to the different impacts considered so, also involved in the decision tree of the choice of midpoint indicators. We also have to choose a "management" MDS, a selection of agricultural-management-practice data to be linked with the soil MDS through models of impact estimation. These input data have to be easily accessible at regional and/or country scales.

Different midpoint categories, corresponding to processes that influence soil quality, will be considered: erosion, effect on soil organic matter (SOM), compaction, biological quality and salinisation. We have focused initial efforts on erosion and effects on SOM because they occur globally and because both erosion (Lal *et al.*, 1999) and organic matter (Fenton *et al.*, 1999) can have major effects on soil quality. To obtain valid estimates of erosion and organic-matter dynamics for a large range of soil types and climates using readily available data, we chose pre-existing simulation models that were both general and simple to parameterise, yet relatively accurate.

2.1. Description of models used

2.1.1 *Erosion: RUSLE*

The RUSLE (Revised Universal Soil Loss Equation) model (Renard and Ferreira, 1993) improves upon the original USLE model. In both models, the fundamental equation is:

$$A = R \times K \times LS \times C \times P \qquad (1)$$

Where A is the computed annual soil loss, R is the rainfall-runoff erosivity factor, K is the soil erodibility factor, LS is a topographic factor combining slope length L and slope steepness S, C is a cover-management factor, and P is a supporting practices factor. Three input databases are required that describe climate, crops and field operations.

2.1.2 *Organic matter: RothC*

RothC (version 26.3) is a simulation model of the dynamics of organic carbon in soil (Coleman and Jenkinson, 2008). The effects of soil type, temperature, moisture content and plant cover are considered in the turnover process. It uses a monthly time-step to calculate total organic carbon (t ha^{-1}) and microbial biomass carbon (t ha^{-1}) on a year to century time-scale. The few inputs it needs are relatively easily to obtain (*e.g.*, mean annual temperature, soil clay percentage, mean monthly rainfall).

2.2. System definition

The system boundary is set at the farm gate. For the moment, estimation of impacts on soil quality does not yet include soils off the farm site, impacted at stages upstream in the supply chain. The temporal boundary covers the inter-crop period (if any just before the crop under consideration) and crop periods, beginning with the start of the inter-crop and finishing with the end of the crop under consideration.

2.3. Functional unit

The method is developed for the assessment of one kilogram of agricultural product, in the present case, grain maize. We chose this crop because it (1) is cultivated globally, (2) requires relatively few crop management activities and (3) often is considered as having more negative environmental impacts than many other crops.

3. Development and Discussion

Tables 1 and 2 show a sample of soil properties (MDS) and agricultural management practices from Brittany, France, and Santa Catarina State, Brazil, selected to assess the impacts of erosion and change in organic matter.

Table 1: Selected soil and climate properties of farm sites in France and Brazil

	FRANCE (Brittany)	BRAZIL (Santa Catarina State)
Soil classification (FAO)	Cambisol	Nitisol
Organic matter (%)	4	3
Clay / Silt / Sand (%)	20 / 48 / 32	62 / 35 / 3
Climate	oceanic	humid sub-tropical
Mean annual temperature (°C)	11	21
Mean annual precipitation (mm)	1060	2200

Table 2: Selected agricultural management practices for farm sites in France and Brazil

	FRANCE (Brittany)	BRAZIL (Santa Catarina State)
Fertilisation	Pig slurry	Pig slurry
Tillage practices	Tillage	Tillage / No-tillage
Crop rotation	Wheat - Maize	Soybean - Maize
Maize crop dates	**Planting**: April - May **Harvesting**: Sept - Oct	**Planting**: Sept - Dec **Harvesting**: Feb - April

Many soil and climate characteristics differ between the two sites. The French soil has a loam / silt-loam texture, while the Brazilian soil has a clay / silty-clay texture, and clay content is a key factor influencing erosion rates. Though mean annual temperatures also differ, the time between planting and harvesting remains approximately the same for maize in Brittany (135-140 days) and Santa Catarina (155 days). Maize is the second most important crop grown in Brazil after soybean, and most of it in Santa Catarina is used for pig production. Likewise, Brittany produces most of the French pork supply; so, organic matter application is in the form of slurry in both cases. Tillage practices are different in France and Brazil. Tillage is representative of France's practices instead of no-tillage for Brazil. Comparison of impacts on soil quality will be made between France with tillage practices and Brazil with tillage and no-tillage.

Our method will be developed by comparing these two contrasting soil-land-climate systems. The method has to be flexible enough to accommodate a variety of soil types and climates, yet remain sensitive to management changes from year to year.

4. Conclusion and perspectives

Agricultural soil impacts off the farm site itself, upstream in the supply chain, will be taken into account in future development of this method to distinguish both direct and indirect components of potential impacts of agricultural activities, including animal production, on soil quality. The units used to express each midpoint impact category are critical, since they have to take into account the susceptibility of the soil to the impact under consideration. Ideally, off-site and on-site impacts could be summed together.

Our approach to assess impacts of agricultural activities on soil quality also is based on the development of a decision tree of the midpoint impact categories to consider when assessing soil quality, as a function of soil and climate conditions. Since contexts can differ according to the product assessed and farm location, it is essential to have a flexible tool to recognise the impact categories involved in that context.

Once developed, the method's impact predictions will be evaluated with sensitivity and uncertainty analyses.

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An LCA of potato production in Ireland: impacts on ecology and environment

Alice D'Arcy[*,1,2], John O'Halloran[1], Colin Sage[2]

1.School of Biological, Earth & Environmental Sciences, University College Cork
2. Department of Geography, University College Cork, Ireland

ABSTRACT

Historical events notwithstanding, the potato maintains a very significant place in the Irish land- and food-scape. The social and demographic impacts emanating from the adoption and subsequent failure of the potato crop in Ireland are manifest. However the past and present environmental and ecological impacts of this system have not been adequately addressed. This paper, through Life Cycle Assessment, examines the potential environmental impacts of contemporary conventional and organic potato production/distribution systems for the first time in Ireland. To establish a baseline as regards ecological impacts, a study of these systems has been undertaken examining the levels of and fluctuations in field level soil properties and macro-invertebrate communities, particularly earthworms, in potato fields, throughout a growing season. These data will be incorporated, if possible, in the LCA, for more comprehensive results, improved comparability, and for the development of specific measures towards increasing the sustainability of both systems in key areas.

Keywords: potato LCA; environmental impacts; soil quality; earthworm diversity.

1. Introduction

Situated within the overall Irish agri-food context, potato production makes up a very small share (<1%) of the Utilizable Agricultural Land; with 12,200ha in conventional production, plus 114ha in certified organic production (DAF, 2006). However as the "*most important field grown horticultural food crop in Ireland*" (Bord Glas, 2001), in addition to consumer preferences for Irish grown high dry matter varieties, potatoes maintain an important position in the Irish diet and agri-foodscape. This agri-foodscape is largely dominated by conventional commercial agri- and horti-culture. The conventional potato system currently consists of just over one thousand registered potato growers and packers (DAFF, 2009). The majority of potatoes grown commercially are sold as unprocessed ware potatoes in the domestic market. There are very few potato processors in Ireland. The sector has contracted hugely over the last number of years and continues to decline in terms of area of production, geographical distribution and numbers of growers (Bord Bia/DAFF 2005). The organic system includes only about 30 registered potato growers. This has been increasing slowly since the beginning of the organic movement in Ireland in 1981 (Organic Europe, 2009). The impacts of these agri-food systems have become increasingly globally-relevant in the contemporary foodscape. Some of the impacts, from GHG emissions to land use and biodiversity effects, have been identified to various extents on international and national levels (O'Sullivan & Gormally, 2002, Mattsson & Wallen, 2003, Hyde *et al.* 2003, Carbon Trust, 2006, O'Brien *et al.* 2008), and there are general assessments of how farming, and potato farming in particular, has affected the Irish environment (Feehan, 2003, D'Arcy, *in prep*). However significant gaps remain regarding the specific environmental and ecological impacts (both positive and negative) of particular crops, agri-food chains, and methods of pro-

[*] Corresponding Author. e-mail: a.darcy@ucc.ie

duction. Life cycle assessment (LCA) is incorporated in this study to measure the impacts of a subset of both conventional and certified organic commercial potato farmers, from production to point of sale, in a particular agricultural region of Ireland (Zone 2a, fig 1). However it has been recognized that there are impact categories not traditionally assessed in LCA, despite their obvious importance in biological systems in particular. Impacts on ecology are one such area for which baseline data, as well as adequate methods of incorporation into LCA, need to be improved (Schenk, 2001, Jeanneret *et al* 2008). To that end an ecological study was undertaken on the same subset of conventional and organic farms addressing the changes in biodiversity and abundance of soil macro-invertebrates, particularly earthworms, and soil properties.

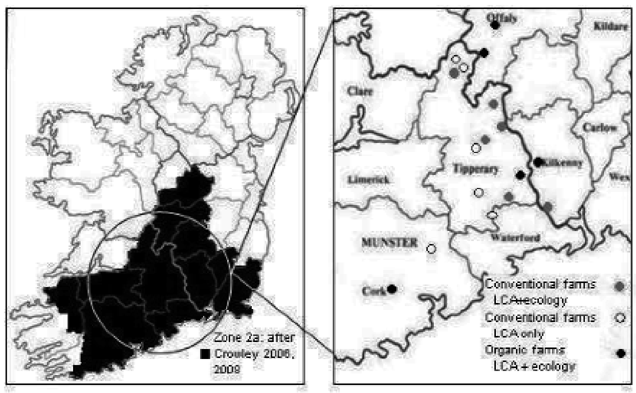

Figure 1: Locations of conventional and organic study

2. Methods

2.1 Life Cycle Assessment methodology

The goal and scope of the project was to carry out an LCA of commercial conventional and organic potatoes in Ireland, from production to point of sale using the functional units of 1kg of ware potatoes for sale for human consumption, and one hectare, for comparative analysis. A series of potato farmers/distributors were approached to participate in providing primary data on inputs and operations throughout an entire growing and distribution season, to the exhaustion of that year's harvest. Conventional farmers were surveyed in 2008/9 and organic farmers in 2009/10. Candidates were selected to cover a representative sample of the range of farm sizes, yields, and distribution methods in the region. Willingness to cooperate was a key selection criterion, as was timing of first field operations for ecological sampling. Eleven conventional farmers participated fully. Cumulatively in the 2008 growing season they cultivated 162ha (~400 acres) of potatoes, ranging from 2.83-81 ha (~7 to 200 acres) per holding, on their own land, on rented land referred to as 'conacre[i]', or both, all of which had been in continuous cultivation for at least ten years. Early and main crop potatoes were

[i] Conacre is a system of letting land for a single cropping season, the lease generally runs from November to November.

grown in a 1-in-4 to 1-in-6 year rotation with cereals and vegetables. Five certified organic farmers participated fully. The combined production area for the 2009/10 season was 9ha (~23 acres), ranging from 0.4-6.5 ha (~1 to 16 acres) per holding. Four out of the five organic farmers grew exclusively on their own land, while one was renting organically certified conacre. Early and main crop potatoes were grown in similar rotations with cereals and vegetables, but rotations of pasture were generally included between successive arable rotations, so that land is not in continuous cultivation for more than 4-6 years. A 'typical' season in both systems includes soil preparation, cultivations, fertilisation, planting, pest control, harvesting, grading, packing, storage and distribution. In a 'typical' season operations begin in early Spring (Feb/Mar) and the 'season' ends the following Spring when potato stores are exhausted. However weather conditions, soil type, and individual behaviour, among other factors, can and do affect the timing, number and duration of operations, as well as yields.

Inputs and operations associated with potato growing, storage and distribution were recorded as the season progressed through a series of semi-structured interviews, site visits, and supported with secondary information from agricultural and other sources. A wide range of distribution methods and markets were utilised by the participating farmers, from 'on-farm' sales, to box schemes, farmers markets, small/medium retail outlets and wholesalers, to the largest wholesale/distribution centres serving the major multiples. The majority of these farmers distributed potatoes themselves to either wholesale or retail outlets, or both. Primary data were used for the majority of the agricultural and distribution phases. Production of inputs including seed potatoes, machinery and infrastructure were included using Ecoinvent (2007) databases. Where detailed information was inaccessible data were inputted based on average data and specific assumptions. Where possible these data will be replaced with data more representative of Irish conditions. For the proportion of crops culled & saved as seed allocation of environmental burdens by mass was applied. The data are currently being processed using Microsoft Excel and GaBi (PE, 2007) with Ecoinvent integrated and LCIA results assigned according to CML (2002) methodology.

2.2 Biodiversity assessment methodology

In order to establish a baseline, and examine the fluctuations in field level soil quality and biodiversity throughout the growing season in field and boundary habitats, an ecological study was carried out on six conventional and five organic farms. A stratified systematic sampling regime was carried out at three general stages in the season;

1/before any field operations (early Spring, prior to initial ploughing),
2/post harvest (after cultivations & harvest complete, late Autumn/Winter),
3/after treatment (the following Spring prior to next crop cultivations)

Three to five replicate soil samples were taken along field and boundary transects at each of the three stages above. Biodiversity and abundance of soil macro-invertebrates , particularly earthworms, were assessed by hand sorting a 25cm^2 area (20cm depth) of soil at each replicate, removing and identifying invertebrates for preservation, identification and analysis according to standard methodologies (Curry *et al*, 2002, Coleman *et al*, 2004, Bartlett *et al*, 2009). Soil properties including soil type, bulk density and total organic carbon were assessed, and soil tests carried out for the farmers by independent laboratories were collected.

3. Results

The collection of LCA and biodiversity data is currently being completed and analysed. Impact categories under consideration include use of resources, energy use, global warming, photo-oxidant formation, acidification, eutrophication, ozone depletion and toxicity. Some results in terms of inputs and impacts per kilogram saleable potatoes and per hectare are shown in table 1 and figures 2 and 3. These results are based partly on proxy data which will be reassessed and/or replaced with data more fitting to Irish conditions in time.

Table 1: Selected farm characteristics; cradle to gate inventory results (not normalised or weighted) functional units are 1kg and 1ha potatoes; plus significant levels of impacts on earthworms.

Table 1	Median values (Q1, Q3)	*Conventional Farmers (n=11)*	*Organic Farmers (n=5)*
Characteristics	ha potatoes farmed	12.95 (4.25, 27.82)	0.83 (0.40, 1.36)
Conventional (2008/9)	t/ha seeding rate	2.47 (2.47, 2.53)	2.47 (2.47, 2.47)
Organic (2009/10)	t/ha yield(s*)	28.1 (22.68, 32.42)	20.86 (16.16, 21.14)
Resource use	kg/kg potatoes	3.68 (2.77, 4.73)	2.13 (1.83, 2.45)
	t/ha	103.40 (78.79, 122.92)	41.45 (33.31, 44.53)
Relative contribution of each life stage to total resource use per kg or /ha (%)	Seed	4.52 (3.96, 8.20)	16.29 (14.36, 21.49)
	Agriculture	67.39 (60.57, 78.98)	48.09 (29.28, 56.36)
	Grading, Packing, Storage	24.16 (7.01, 28.91)	31.23 (17.05, 33.84)
	Distribution	0.80 (0.47, 2.08)	6.53 (3.53, 14.41)
Net Energy Resources	MJ/kg	4.00 (3.80, 4.90)	4.80 (3.73, 5.66)
	MJ/ha	109787.55 (104913.99, 129015.50)	85842.58 (77546.36, 85858.72)
Potential impacts: GWP	kg CO2equlv /kg	0.14 (0.14, 0.19)	0.27 (0.22, 0.28)
	t CO2equlv /ha	4.24 (3.97, 5.05)	4.32 (4.07, 4.78)
Relative contribution of each life stage to total GWP per kg or per ha (%)	Seed	-2.21 (-3.57, -1.89)	17.53 (-3.41, 17.61)
	Agriculture	90.68 (88.57, 94.99)	73.63 (61.64, 78.05)
	Grading, Packing, Storage	2.70 (1.05, 6.46)	0.03 (-0.67, 0.69)
	Distribution	6.25 (3.21, 10.24)	20.14 (9.43, 31.17)
Significance of impacts of farming on earthworms	on Abundance (per m^2)	ξ^2 = 40.932; df=2; p=0.00	ξ^2 = 6.672; df=2; p=0.036
	on Live biomass (per m^2)	ξ2 = 37.117; df=2; p=0.00	ξ^2 = 3.780; df=2; p=0.151

A high degree of variation both within and between systems was recorded. The resource use per kg and per ha was consistently higher in conventional systems, as was the relative contribution to resource use from the agricultural production stage. Energy use per kg was higher in organic systems, but lower per ha. Impacts were characterised using CML (2002). Global warming potential was the highest impact category in both systems. Median potential emissions values (tCO2eq) per ha were similar in conventional and organic systems, but higher per kg in organic systems (fig 2). Further LCIA results are shown in figure 3. The range of total organic carbon levels in soils was very variable, ranging from 4-35% and a high degree of variation in TOC levels was measured between the start and end of the season, but overall there was a median decrease of 10-11% in both systems. In conventional systems differences between pre-treatment, post harvest and 'recovery' samples in both earthworm abundance and live biomass levels (per m^2) were significant at the $P<0.01$ level. In organic systems the response was more varied through time, with no significant difference in abundance or biomass at the $P<0.01$ level between stages.

4. Discussion

Yields per hectare appear to be a key determining factor in the potential impacts from production to point of sale per kg. Potential benefits in organic production systems such as lower input levels of mineral fertilisers and pesticides, are being lost due to lower yields, higher relative impacts of distribution systems and energy use. However per hectare 5 out of 6 median impact potentials shown (fig 3), were lower in organic systems. Ecological results show that conventional farm operations significantly negatively affected earthworm abundance and biomass, with little or no recovery the following Spring, whereas farm operations in organic systems did not. TOC results require further analysis due to the high variability in both initial levels and changes throughout the season. However it appears that there may be substantial trade-offs to consider between conventional and organic farming systems as regards environmental and ecological impacts. Results of both systems can be regarded as worst case scenarios due to particularly difficult weather conditions in the harvest seasons in both years leading to substantial losses in many cases.

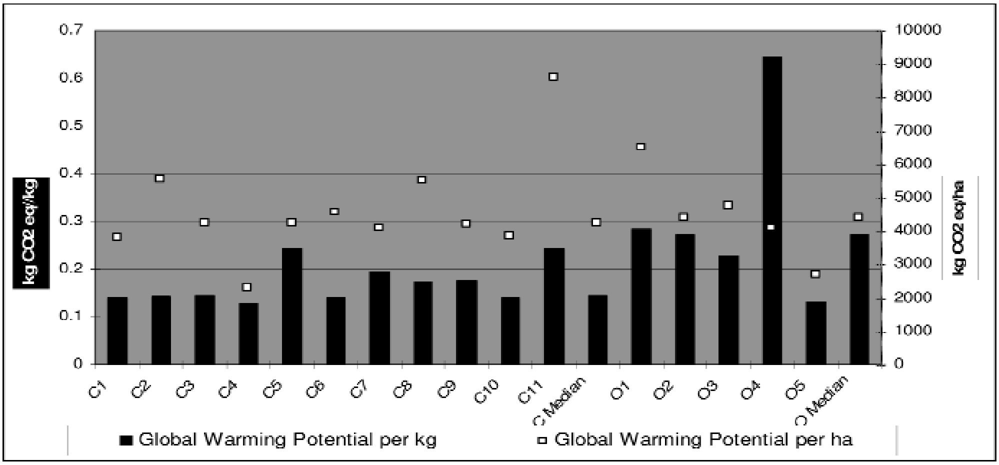

Figure 2: Global warming potential per kg and per ha for conventional & organic systems (C = conventional, O = organic, plus median values for both)

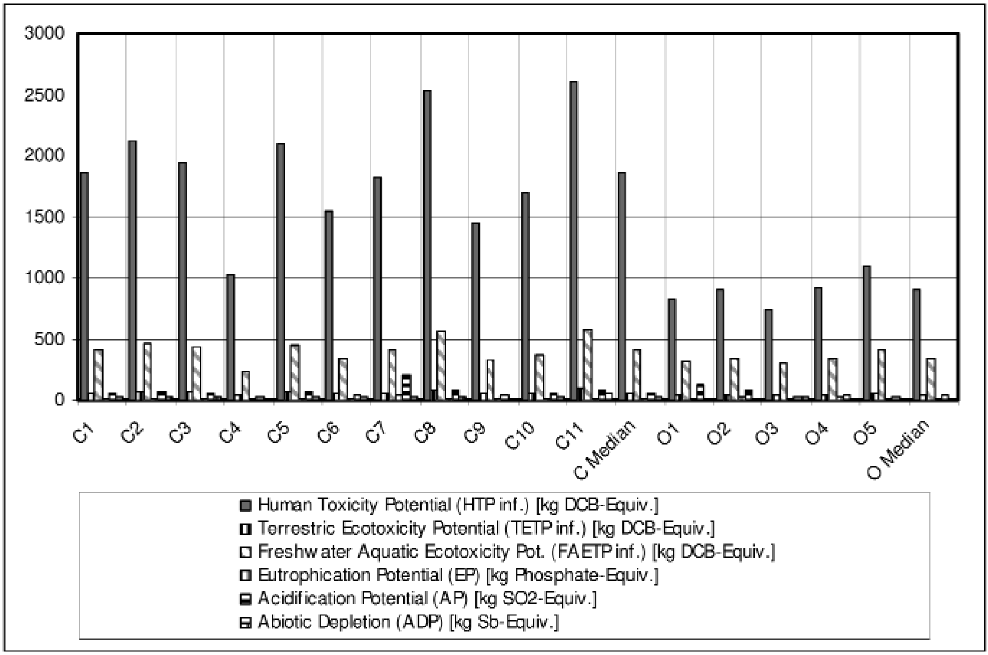

Figure 3: Other potential impacts of conventional & organic systems per ha (CML 2001-2007)

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Validation of a method for biodiversity assessment in LCA (SALCA-Biodiversity) using indicator species groups

Bärbel Koch[1,2*], Philippe Jeanneret[1], Daniel Baumgartner[1], Thomas Walter[1] & Gérard Gaillard[1]

[1]Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zürich, Switzerland
[2]ETH Swiss Federal Institute of Technology, CH-8044 Zürich, Switzerland

ABSTRACT

The SALCA-Biodiversity (SALCA-BD) method developed by ART with the aim to integrate biodiversity as an impact category for agricultural production in LCA was validated for two of the eleven indicators: grassland flora and grasshoppers. On the basis of management practices of ten farms grassland plots directly recorded by the farmers, biodiversity scores were calculated with SALCA-BD. Grassland flora and grasshopper field data recorded *in situ* were compared to the calculated scores at the plot as well as at the farm level. Significant correlations at the plot level were found between calculated scores and field data for both grassland flora and grasshoppers. At farm level significant correlations were found for the grassland flora only. The results show that SALCA-BD method is appropriate for estimating the impact of management practices on indicator species groups and shows the wished sensitiveness with regard to different intensities of agricultural land use.

Keywords: LCA, biodiversity, flora, grasshoppers, agriculture

1. Introduction

In the context of Life Cycle Assessment for agriculture, we developed a method for the integration of biodiversity (species diversity) as an impact category, SALCA-Biodiversity (Swiss Agricultural Life Cycle Assessment for Biodiversity) (Jeanneret *et al.*, 2006; Jeanneret *et al.*, 2008). This method aims at assessing along a midpoint approach the impact of farming operations, management systems and farms on biodiversity in a predictive manner.

Biodiversity in the broadest sense of the Rio Convention cannot be totally measured and a single indicator is unlikely to be devised even in agro-ecosystems (e.g. Büchs, 2003). Instead, groups of indicators should be selected that are sensitive to the environmental conditions resulting from land use and agricultural practices, and give as representative a picture as possible of biodiversity as a whole. We selected indicator species groups (ISGs) according to their linking to agricultural activities, their association to specific habitats and their place in the food chain (Jeanneret *et al.*, 2006): flowering plants, birds, small mammals, amphibians, snails, spiders, carabid beetles, butterflies, wild bees, and grasshoppers. The impact assessment distinguishes between the overall species diversity (OSD) of each ISG, and the diversity of the ecologically demanding species or/and stenotopic species (EDS). To assess the impact of agricultural practices on the selected indicator species groups, inventory data reflecting detailed management options were specified (e.g. quantity of fertilizers, number of cuts). Based on information from literature and expert knowledge, a scoring system was developed that estimates the response of every ISG to the management options taking into ac-

* Corresponding Author. e-mail: baerbel.koch@art.admin.ch

count the habitat where they take place (e.g. grasslands, cereals, semi-natural habitats). Scores of management options were then aggregated at the field level (e.g. fertilization and cutting regime) in order to compare agricultural systems. The impact of land use on biodiversity at farm level was calculated by further aggregating the biodiversity scores obtained at field level under consideration of the ecological relevance of the habitats concerned.

The aim of this study is to compare outputs (scores) from SALCA-BD calculated with management data with field data recorded *in situ* for grassland at the plot and farm level. Two of the ISGs, vascular plants and grasshoppers, were chosen for this comparison for following reasons: Vascular plants correlate well to the overall biodiversity of a region (Duelli *et al.*, 1998) and grasshoppers are typical grassland insects, 80% of the species in Switzerland being able to grow on meadows and pastures (Schneider *et al.*, 2001). Both indicator groups are relatively easy to record and identify, and react sensitively to management practices (e.g. Marini *et al.*, 2008).

Following question was addressed: Does the SALCA-BD scores for vascular plants and grasshoppers correlate with the respective data recorded in the field at both plot and farm levels?

2. Methods

In 2008, ten grassland dominated farms were chosen along a management intensity gradient at the southern margins of the Swiss Jura Mountains (Canton of Aargau) at altitudes between 350 and 750 m a.s.l. Vascular plants and grasshoppers were recorded in the field on every grassland plot of ten (n=198) resp. six (n=77) farms. Data on agricultural practices regarding fertilisation, mowing, grazing and weed or mice control that have taken place in 2008 at every single plot were directly obtained from the farmers.

All plant species present on a 25 m^2 circle representative for the plot were recorded in the field. In case of a heterogeneous plot presenting a mosaic of patches of different vegetation types, a plant list and the percentage of area covered by every patch were also recorded. Grasshopper species were recorded visually and acoustically on sunny days with little or no wind during one hour walk through the plot. From the field data species richness and high nature value scores based on species composition were derived. High nature value scores for the vascular plant group were obtained with a point system for valuable species to the Swiss Ecological Quality Ordinance (EQO) (BLW, 2008b; 2008a) and the UZL plant lists (BAFU & BLW, 2008). Species not mentioned in the list received zero points. To calculate the total plant species richness and the high nature value scores of heterogeneous plots a weighted average was performed taking into consideration the percentage of area covered by each vegetation patch. High nature values for the grasshopper group were derived from the Swiss Red List (RL) for grasshoppers (Monnerat *et al.*, 2007) and the UZL grasshopper list (BAFU & BLW, 2008). To calculate high nature value scores points were assigned depending to the level of high nature value and endangerment of the species. Because of the restricted number of grasshopper species with high nature value mentioned in the RL and UZL lists, the majority of plots resulted in a score of zero points. Therefore a minimum of one point was attributed to every grasshopper species and RL scores were summed to UZL scores resulting in a single high nature value (RL+UZL score) for the grasshopper group.

SALCA-BD outputs, OSD and EDS (grasshoppers only) scores, calculated on the basis of agricultural practices, were compared to the species richness and high nature values for both plant and grasshopper groups (Tab. 1). With the statistical program R (R Development Core Team, 2008) significant correlations at the plot level were tested with the Spearman's rank correlation test and at the farm level with the Pearson's product-moment correlation test.

Table 1: SALCA-BD scores and field data pairs compared. OSD = Overall species diversity; EDS = Ecologically demanding species.

Grassland flora			Grasshoppers		
SALCA-BD scores		Field data	SALCA-BD scores		Field data
Grassland flora OSD	↔	Species richness	Grasshopper OSD	↔	Species richness
Grassland flora OSD	↔	UZL score	Grasshopper EDS	↔	Species richness
Grassland flora OSD	↔	EQO score	Grasshopper OSD	↔	RL+UZL score
			Grasshopper EDS	↔	RL+UZL score

3. Results

Overall 294 plant and 17 grasshopper species were recorded with an average per plot of 29 plant resp. 6 grasshopper species.

Table 2 summarizes the results of the correlations between SALCA-BD scores (OSD and EDS) and field data recorded *in situ* (species richness, UZL score, EQO score and RL+UZL score) for grassland flora and grasshopper groups at the plot and farm level.

Table 2: Results of the correlation tests. P-value: * = < 0.05; ** = < 0.01; *** = < 0.001.

		SALCA-BD score – field data	Correlation value
Grassland flora	Plot level (N=198)	OSD - Species richness	0.578***
		OSD - UZL score	0.624***
		OSD - EQO score	0.609***
	Farm level (N=10)	OSD - Species richness	0.735*
		OSD - UZL score	0.734*
		OSD - EQO score	0.755**
Grasshoppers	Plot level (N=77)	OSD – Species richness	0.389***
		EDS – Species richness	0.361**
		OSD – RL+UZL score	0.338**
		EDS – RL+UZL score	0.323**
	Farm level (N=6)	OSD – Species richness	0.658
		EDS – Species richness	0.696
		OSD – RL+UZL score	0.583
		EDS – RL+UZL score	0.628

Correlations between SALCA-BD scores and species richness or high nature values for grassland flora and grasshopper at the plot level were overall positively correlated. The correlation values were higher for the grassland flora than for grasshoppers. For grassland flora the highest correlations between SALCA-BD score and field data were found for the UZL score, one of the two scores suggesting high nature value of species composition. For grasshoppers, in contrast, the highest correlation values with SALCA-BD scores, OSD and EDS, were found for species richness. Correlations of field data with the OSD scores resulted to be higher than with the EDS scores (only grasshoppers).

At the farm level, correlations between the SALCA-BD scores and field data recorded *in situ* resulted to be significant only for grassland flora. The highest correlation value was found between the farm OSD and the farm EQO scores. Despite the relatively high correlation values at the farm level for grasshoppers, both OSD and EDS scores resulted to be non significantly correlated with the species richness and the RL+UZL score.

4. Discussion

Cultivated land is used as habitat by numerous plant and animal species, and agricultural practices have a major impact on the biodiversity of this environment (e.g. Stoate *et al.*, 2001; Benton *et al.*, 2002; Robinson *et al.*, 2002). Appropriate monitoring methods to evaluate and reduce the impact of agricultural farms on biodiversity are needed. SALCA-BD is an indirect method which enables to assess biodiversity of a farm, plot or crop in a cheap, fast and simple way (Jeanneret *et al.*, 2006).

At the plot level, the significant correlations for both grassland flora and grasshopper indicator groups between calculated scores and field data shows that the SALCA-BD method is appropriate for estimating the impact of management practices on indicator species groups, at least the ones investigated in this study. These results suggest the validation of the scoring system, based on results presented in the scientific literature and expert knowledge, and in particular SALCA-BD aggregation steps at the plot level. Aggregated plot scores at farm level conducted to positive significant correlations with *in situ* observations for vascular plants but not for grasshoppers, although positive but not significant for the latest.

Correlation values for the grassland flora were overall higher than for the grasshopper indicator, which may be due to the smaller number of plots recorded, the fewer species of grasshoppers compared to that of grassland flora and/or plot heterogeneity which was taken into consideration only for the flora group, since it was assumed that grasshopper species were moving freely within the plot. In addition, an important feature observed in grasshoppers was the high impact of the surrounding land use, reported also for various other insect groups (Duelli *et al.*, 1999; Jeanneret *et al.*, 2003), and not taken into account in SALCA-BD. Surrounding areas indeed can positively or negatively affect biodiversity (e.g. De Snoo *et al.*, 1999; Tscharntke *et al.*, 2005). The history of the plot, non considered in SALCA-BD, can play an important role too (Smith *et al.*, 2003; Marriott *et al.*, 2004). The inexactness of the estimations due to both abovementioned limitations of the method affect the results, but for the plot level it was shown that even with such constraints, SALCA-BD sensitivity was high enough to lead to significant correlations between calculated scores and field data. At the farm level, good results were achieved only for the grassland flora. However, the relatively high correlation values between grasshopper SALCA-BD scores and field data at the farm level give evidence that not the data but the few farms recorded (N=6) is the probable reason for the undetected significance.

For the grassland flora, the highest correlation values with SALCA-BD scores were found for the high nature value scores: at the plot level with UZL score and at the farm level with EQO score. Plants mentioned in these two lists are species specific to cultivated land. In contrast, no distinction between cultivated land and forest plant species, these latter encountered in plots at the forest edge, was done for the record of the species richness. The focus of SALCA-BD on agricultural habitats could explain the higher correlations with the high nature value scores mentioning species growing on cultivated land only and the lower correlation values with the species richness often including also species unspecific for this habitat.

For the grasshopper group, both SALCA-BD scores (OSD and EDS) were higher correlated with the species richness. Because of the limited number of grasshopper species with high nature value the RL score and UZL score were summed, which may have lead to an inadequate point system to represent the high nature value score leading to lower correlation values with SALCA-BD scores. Correlations between the OSD scores and both the species richness and the quality value for grasshoppers were higher than with the EDS scores, showing a higher sensitivity of the method when calculating OSD values.

The results obtained for grassland flora and grasshoppers cannot be directly transferred to the other indicators. However, the scoring system of the remaining groups was established with the same method, i.e. based on scientific literature and expert knowledge. There are therefore good prospects that SALCA-BD gives satisfying results also for the indicators not validated in this study.

The study presents the high sensitivity of the method concerning the impact of different agricultural management practices on biodiversity at the plot level. SALCA-BD resulted to be a suitable method to investigate the optimization of agricultural management activities as well as the comparison of farms or different land uses relative to biodiversity. At the farm level sensitivity with regard to agricultural practices was attained only for grassland flora; the outcome for grasshoppers possibly affected by the few replicates should be ascertained with more research.

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Contribution to modelling soil erosion and water consumption in life cycle assessment of agricultural systems

Montse Núñez[1,*], Assumpció Antón[1,2], Pere Muñoz[1], Joan Rieradevall[3,4], Juan Carrasco[5]

[1]IRTA, SosteniPrA. Ctra. de Cabrils, Km 2. 08348 Cabrils, Barcelona, Spain.
[2]Departament d'Enginyeria Quimica. Universitat Rovira i Virgili (URV). 43003 Tarragona, Spain.
[3]ICTA, SosteniPrA. Institute of Environmental Science and Technology (ICTA), Universitat Autònoma de Barcelona (UAB). 08193 Bellaterra, Barcelona, Spain.
[4]Chemical Engineering Department. Universitat Autònoma de Barcelona (UAB). 08193 Bellaterra, Barcelona, Spain.
[5]Research Centre for Energy, Environment and Technology (CIEMAT). Avda Complutense, 22. 28040 Madrid, Spain.

ABSTRACT

In spite of the importance of soil and water for agricultural activities, life cycle assessment (LCA) still lacks agreed methods for assessing impacts due to land and water use. Neglecting soil and water impact categories in LCA can be particularly serious in arid and semi-arid areas such as Spain, where both soil and water are limited natural resources. In this study, we contribute to the further development of these impact categories traditionally lacking in LCA, focusing on off-stream water consumption and soil erosion potential impacts, considering the life cycle inventory (LCI) and the life cycle impact assessment (LCIA) stages of LCA. The methodology was tested on plots of agricultural production land located in the Spanish water basins. Spatial optimization for planting the crop rotations analysed, in terms of soil and water use, could be achieved by considering outcomes from the regionalized LCA.

Keywords: Energy crops, Land use, Life cycle impact assessment (LCIA), Water use

1. Introduction

Stress on global water resources and soil reserves are internationally recognized as a burning issue which must be addressed by national environmental agendas. The sustainable management of soil and water is a priority when they are intensively used, for example in agriculture, and also in arid and semi-arid regions, such as Spain. In this framework, the increasing cultivation of energy crops in Europe is likely to have a major effect regarding future environmental pressures on the soil and water reserves of each country.

Today, although soil and water are gaining importance in life cycle assessment (LCA) methodology, these are not yet well established impact categories, and can only be partially addressed in LCA studies. The most common approach for assessing soil and water impacts in LCA is by an inventory of the quantity of water used (m^3) and the soil occupied (m^2) for the development of the activity, both expressed in terms of the inventory flow. Information on many important parameters linked to the quality, origin and fate of the resource is not included, leading to at least incomplete, and at worse, erroneous results.

Given the lack of agreed LCA methodologies to deal with soil and water impacts and the forecast for a rise in energy crop production in Spain, the Spanish Ministry of Science and

[*] Corresponding Author. e-mail: Montserrat.nunez@irta.cat

Innovation is funding an in-depth study[1] focusing on the suitability of growing energy crops in the country, with one of the aims being to improve current soil and water LCA characterisation methods.

The objectives of this study were threefold:

- Improving LCA methods to evaluate off-stream water consumption impacts. Off-stream water consumption refers to use of water out of the water body, where water, after being used, leaves the system through evaporation, is incorporated into the product, is transferred to a different water basin or discharged into the sea (Owens, 2001). Improvements of the methodology concern both the inventory (LCI) and the impact assessment (LCIA) steps.
- Improving LCA methods to incorporate soil erosion potential impacts in the inventory and impact assessment stages.
- Identifying appropriate production areas and energy crop rotations to minimise the environmental effects of water consumption and soil erosion for growing energy crops in Spain, using the proposed inventory and impact assessment schemes.

2. Material and methods

2.1. Crop rotations

Impacts of water consumption and soil erosion were studied on a crop rotation basis, instead of using the single crop period as the length of time for the elementary flow. Farmers cultivate crops in a rotation system, taking crop functionalities into account (Kägi *et al.*, 2007; Nemecek *et al.*, 2008). Also management practices implemented during the cultivation of a crop (e.g., application of fertilizers) may benefit subsequent crops. Thus, we did not allocated impacts of water consumption and erosion to single crops of the rotation, as it may lead to misleading results. We analysed two rotations with energy crops and a reference rotation without them. The crops are destined for food or energy purposes. The systems studied so far are as follows, with the energy crop of the bioenergy rotations underlined:

- Winter barley – winter wheat – unseeded fallow (B-W-F): cereal crop rotation of 2 years with the soil lying fallow in the last year. This rainfed rotation, very common until recently in Spain, was selected as the reference agricultural system to compare soil losses and water consumption impacts against those of rotations with energy crops. Both water consumption and soil erosion impacts were evaluated for this reference rotation system.
- Winter barley – winter wheat – <u>oilseed rape</u> (B-W-R): rainfed cereal crop rotation of 2 years with a bioenergy crop in the last year. The impact assessment was performed for water consumption and erosion impacts.
- <u>Poplar – poplar – poplar</u> (P-P-P): short rotation coppice, with a life-span of 15 years, with five consecutive cycles of 3 years (five cuts). It is a deficit-irrigation rotation (i.e., watered below water requirements of the plant). The assessment was carried out only for water consumption impacts.

Water consumption was estimated in 117 agricultural plots scattered throughout the country. Soil erosion was estimated in 55 selected plots.

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[1] *Singular and Strategic Project for the development, demonstration and evaluation of the viability of the commercial production of energy from dedicated crops in Spain* (*SSP On Cultivos*, http:www.oncultivos.es).

2.2. LCI and LCIA methodologies

2.2.1. Water assessment

The total water consumption of each rotation was estimated for the LCI, considering water coming from irrigation (blue water) and the uptake of soil moisture (green water). Crop evapotranspiration was calculated using the FAO approach (Allen *et al.*, 1998), adjusting crop evapotranspiration values to water stress conditions. Soil moisture availability, i.e., effective precipitation, was calculated using the runoff curve number method adapted to the Spanish conditions (Ferrer, 1993; MOPU, 1990). The reference flow for water consumption impacts was 1 m^2 of agricultural plot cultivated during a 3-year rotation.

For the LCIA, the off-stream water consumption impact method of Pfister *et al.* (2009) was used. These authors developed endpoint characterisation factors compatible with the Ecoindicator 99 framework on a water basin scale for the areas of protection resource depletion, ecosystem quality and human health. At the midpoint level, they also proposed a water deprivation impact category. Impacts of blue water consumption due to growth of the selected energy crop rotations were evaluated using these midpoint and endpoint approaches, while green water was only assessed with midpoint factors. In addition to the life cycle oriented approach, the green water was assessed by measuring a local indicator of the climatic aridity in which crops were grown. This indicator, defined by the Water Footprint Network (http://www.waterfootprint.org) and applied in Núñez *et al.* (submitted), measures the relation between the available soil moisture and the green water consumed by the crop.

Geographical information systems (GIS) were used to obtain local data on water consumption for the LCI stage and to spatially represent the results of the impacts (Núñez *et al.*, 2010).

2.2.2. Erosion assessment

The follow-up LCI data has to be gathered:

- Soil losses (kg m^{-2} $rotation^{-1}$) caused by the evaluated land use activity at plot level. They can be measured using the universal soil loss equation (USLE, Wischmeier and Smith, 1987) or other estimation models.
- Bulk density of the topsoil (kg m^{-3}), as it is the upper layer of soil, thus the first to be eroded.
- Area size ($m^{2)}$ and duration (years or number of rotations, depending on the time unit of the functional unit) of land use occupation. In this study, impacts were calculated for an area of 1 m^2 and a one 3-year-rotation of land use occupation.
- Location (coordinates).

For the LCIA stage, we proposed an endpoint indicator for the impact pathway of resource depletion (soil as a resource). In the cause-effect chain, current loss of topsoil due to the growth of crop rotations (environmental intervention) restricts the ability of the topsoil to sustain future uses (environmental consequence). Because the effect of land use relies on site specific conditions, impacts are presented relative to biogeographical conditions. Regionalized characterisation factors were therefore differentiated, based on the local available soil reserves (FAO, 2007).

Here, apart from using GIS to gather primary data for the LCI and to represent scores of the land use impacts, we used geospatial tools to derive the necessary impact factors.

The results of the (blue and green) water and soil erosion assessments can be used to select the most appropriate locations and rotations to minimise environmental impacts of water consumption and land use for growing energy crops in Spain.

3. Results and discussion

3.1. LCI of water consumption and soil erosion

The results of the (blue and green) water consumption and soil erosion regionalized at basin level for some of the main watersheds in the country are shown in Table 1.

The rotations with oilseed rape (B-W-R) and fallow (B-W-F) are rainfed rotations which do not have blue water consumption. Looking at green water consumption, the rotation with oilseed rape had the higher water requirements. As for poplar rotation, it consumes less soil moisture than the reference system (B-W-F) in some watersheds (e.g., Duero), due to the higher water requirements of wheat and barley in the winter months and the distribution of rainfall and potential evapotranspiration during the year.

The results for soil erosion show that the rotation with fallow (B-W-F) had more soil losses, due to the soil being more prone to erosion during the last year (F). Soil losses from the poplar rotation have yet to be assessed, but we consider they are unlikely to be of great importance, as the soil is protected from erosion with adventitious vegetation after the first months of crop planting.

Table 1: Water consumption and soil losses of the rotations studied in some of the main water basins in Spain.

Water basin	Region	Crop rotation	Blue water consumption (m^3/m^2 rotation)	Green water consumption (m^3/m^2 rotation)	Soil erosion (kg/m^2 rotation)
Ebro	North	B-W-F	0	0.795	0.500
		B-W-R	0	0.883	0.272
		P-P-P	0.900	0.812	n.a.
Guadalquivir	South	B-W-F	0	0.694	0.743
		B-W-R	0	0.755	0.443
		P-P-P	0.846	0.679	n.a.
Duero	North	B-W-F	0	0.709	0.339
		B-W-R	0	0.771	0.174
		P-P-P	0.900	0.688	n.a.
Segura	Southeast	B-W-F	0	0.632	1.381
		B-W-R	0	0.670	0.871
		P-P-P	0.900	0.652	n.a.

B: winter barley; W: winter wheat; F: fallow; R: oilseed rape; P: poplar.
n.a.: not assessed.

3.2. LCIA of water consumption and soil losses

Figure 1 shows a summary of the impact results for water consumption and soil erosion assessments. This figure shows how water and soil erosion impacts vary between water basins. While watersheds in the north and north-east of the country scored well for the rotation with irrigation (P-P-P), this rotation had high impacts in some basins in the south and south-east (Fig. 1a). The green water assessment, especially useful for the evaluation of rainfed rotations, shows that water basins in the north-east of the country are also a good location for minimizing impacts from green water consumption (Fig. 1b). However, these north-eastern basins scored the worst for erosion impacts (Fig. 1c). Fig. 1a and 1c also show that some of the basins with the highest impacts for blue water consumption were appropriate for reducing impacts from soil losses.

The uneven distribution of water consumption and soil erosion damage in the water basins cannot be analysed by a country-differentiation of impact factors. These regionalized results show that a single watershed is not capable of simultaneously minimizing water consumption and soil erosion impacts in the cultivation of the analysed rotations. The best spatial distribution depends on the priority resource to be protected.

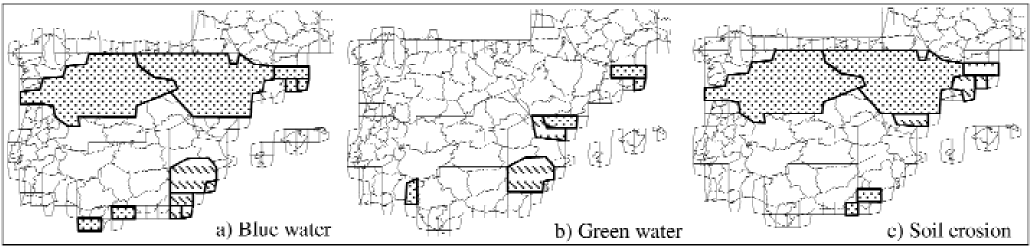

Figure 1: more appropriate (dotted) and less appropriate (lined) for planting the analysed energy crop rotations regarding impact assessments for blue (a) and green (b) water consumption and for soil erosion (c)

4. Conclusions

In this research, we have made a contribution to further develop the impact categories of water consumption and soil erosion, traditionally lacking in life cycle assessment studies. The proposed framework was applied to agriculture rotations with energy crops, but it is suitable for all types of agricultural systems. Our assessment shows that there is no a specific studied crop rotation cultivated in a specific watershed which is capable of minimizing both blue and green water consumption, as well as soil erosion impacts, at the same time.

Further research is focused on this water consumption and soil erosion evaluation for more energy crop rotations whose technical, economical and environmental viability in Spain is currently under study by the Singular and Strategic Project On Cultivos.

Research is also planned to map water and erosion impacts in function of the geographical aptitude, by overlapping the LCA results with edaphic and climate conditions (e.g., minimum and maximum temperatures, rainfall) at the level of agrarian regions.

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