Integrated assessment of agro-ecological systems: The case study of the "Alta Murgia" National park in Italy

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ABSTRACT

Several indicators and methods have been already applied for sustainability assessment in agriculture. The links between sustainability indicators, agricultural management and policies are not well explained (Wei et al., 2009). The aim of this study is to combine biophysical and monetary sustainability assessment tools to support agriculture policy decision-making. Three methodological steps are considered: i) the environmental impacts of farms are assessed using terrestrial acidification, freshwater eutrophication, soil and freshwater ecotoxicity as well as natural land transformation; ii) the most relevant indicators of agriculture damages on ecosystems quality are aggregated into an index; iii) the farm index score is combined with farm assets, land and labour, into the Sustainable Value approach (SVA), as indicator of natural resources used by farms. The methodology was applied in a case study on arable farms with and without animal husbandry of the "Alta Murgia" National Park. The sampled crop farms have a higher sustainable value using their economic and environmental resources. Mixed farms need to improve their resource use efficiency. Although crop farms have lower land-use efficiency than mixed farms, our results suggest, that specialized crops farms generally perform better in terms of ecosystems quality preservation. Finally, we find that Life Cycle Assessment (LCA) providing a measure the environmental impacts of farms clearly enriches the SVA.

1. Introduction

Sustainability assessment is considered an important step towards sustainable human activities (Pope et al., 2004). Scientists have developed several different sustainability evaluation tools in the last thirty years such as biophysical, monetary tools and sustainability indicators to deal with the triple bottom dimensions of sustainability (environmental, economic and social) (Gasparatos and Scolobig, 2012; Van Passel et Meul, 2012; Kloepffer et al., 2008). Interesting reviews of different approaches for sustainability assessment can be found in Neumayer (2003), Gasparatos et al., (2008), Jeswani et al. (2010) and Van Passel and Meul (2012). However, the scientific debate between supporters of monetary or biophysical tools remains unsolved (Gasparatos and Scolobig 2012). Moreover, biophysical and monetary assessment methods differ also in their basic concept of value, relying on cost of production and utility theories of value respectively (Gasparatos et al., 2009). According to Gasparatos et al. (2009), sustainability assessment based on only monetary or biophysical tools ignores the interaction between the two different approaches resulting in a great deterioration of the decision making process. The combination of biophysical and monetary tools may help to achieve a wider sustainability perspective. These "hybrid approaches" (Gasparatos et al., 2008) were strongly fostered in order to balance the simplicity, the wider acceptance and the easy communication characteristics of monetary tools with the more strict and objective relation with ecosystem functions and flows of the biophysical ones, with a logical effect on the improvement of systems' sustainability. In order to avoid critical issues related to consistency and weighting between environmental, economic and societal priorities (Hoogmartens et al., 2014), monetary and biophysical sustainability assessment approaches could help to provide decision makers tools for a simplified and standardized sustainability assessment (Jeswani et al., 2010).

Several indicators such as water withdrawal, threatened species, soil organic carbon content, soil nutrient retention capacity, fertilizers and pesticides use, etc. (Reytar, et al. 2014) were developed to understand the complex relationships between agriculture and environment, but links between sustainability indicators and agricultural management are not well explained (Wei et al., 2009).

The aim of this study is to cover this deficiency by exploring options for combining biophysical a monetary sustainability assessment tools to support agriculture policies decision-making at local, regional or national level. To achieve this goal, the Life Cycle Assessment (LCA) methodology (biophysical tool) was integrated into a monetary sustainability assessment tool: the Sustainable Value Approach (SVA). LCA has been used to define the environmental impacts of the agricultural activities at the farm level, while SVA allows local policy makers to compare the sustainability performances of different farm management strategies. The proposed methodology was applied in a case study to the agricultural system of the "Alta Murgia" National Park (hereinafter simply referred as Park). According to the EC Reg. 1242/2008 establishing a Community typology for agricultural holdings, the typologies of agricultural holdings inside the Park are: mixed crops-livestock; specialist field crops and specialist grazing livestock (Ente Parco Nazionale dell' Alta Murgia, 2010).

This study was a cradle-to-farm gate study, in which all the raw materials and processes take place form raw material extraction or production, till crops or livestock production.

This paper addresses the following research questions: a) is it possible to combine biophysical and monetary sustainability assessment tools in a meaningful and consistent way to agro-ecosystems?; b) is this methodology suitable for investigating structure policy measures to improve the sustainability of agriculture in natural areas?

The paper is structured as follows. Section 2 focuses on the logical framework and the methodologies used in the assessment of environmental and socio-economic impacts of the farm activities inside the Park. In Section 3 the main results are presented. The paper concludes with a discussion and the conclusions (Section 5).

2. A pathway to a more integrated sustainability assessment

2.3. An integrated sustainability assessment of agro-ecosystems

Agro-ecosystems are arguably the most managed ecosystem in the world (Stoorvogel et al. 2004; Wei et al., 2009). In the past, agro-ecosystems were managed and evaluated overemphasizing their social and economic components (Wei et al., 2009). According to different authors, this has caused many changes to ago-ecosystems like land degradation, loss of biodiversity, groundwater depletion, greenhouse gas emissions and erosion (Conway, 1985; van der Werf and Petit, 2002; Dale and Polansky, 2007). The increasing concern about the negative impacts of agricultural activities on natural resources underlies the development of many methods for their evaluation (for a thorough review sees van der Werf and Pertit, 2002; Payraudeau and van der Werf, 2005). In this context, sustainable agriculture can be defined as the management of agro-ecosystem in such a way that it can maintain its biological diversity, productivity and regeneration capacity today and for the future (Van Cauwenbergh et al. 2007). In more detail, Pretty et al. (2008) defined agriculture sustainability as the capability of agricultural systems to: (i) integrate biological and ecological processes, (ii) minimize the human-made inputs, and (iii) make productive use of farmers' knowledge and their collective capabilities. Several models integrate biophysical and economic assessment of agro-ecosystems sustainability (for a thorough review see Janssen and van Ittersum, 2007). Stoorvoegel et al. (2004) propose with the so-called Trade-off Analysis Model an integrated biophysical and economic approach for assessing sustainability of agro-ecosystems, highlighting the role of temporal and spatial scales to supply policy-makers with useful indicators. Wei et al. (2009) used the force-pressure-state-impact-response approach to identify the interactions between biophysical and economic models in order to provide a comprehensive evaluation of farm's performance. Paracchini et al. (2015) presented another approach to sustainability assessment at different spatial level (single farm, farming region, etc.) in combination with a wide range of indicators. According to Dantsis et al. (2010), the application of multiple criteria in agricultural system sustainability assessment requires several assumptions and simplifications although it has also several advantages (e.g. representation of the current agricultural management practices, the simplification of the composite concept and its applicability to different spatial scales). An interesting evaluation of the pros and cons of aggregate indicators for agricultural sustainability assessment is given by Gomez-Limon and

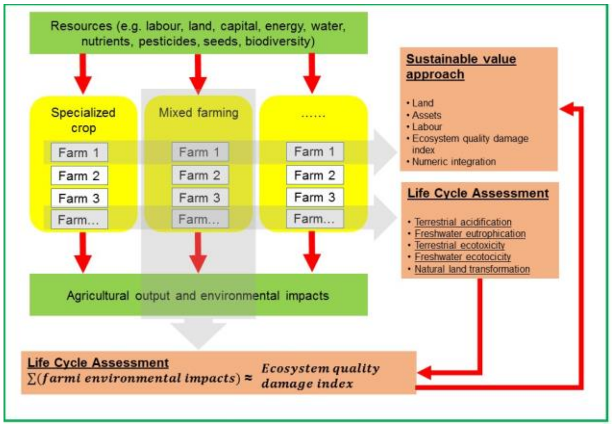
Sanchez-Fernandez (2010). Usually, these "indicator lists" (Gasparatos et al., 2009) have been developed in order to capture sustainability issues relevant for a specific context. Therefore, they are not widely applicable. One example of this approach is the project "Agroecosistemi" [1] supported by the Park. This approach is based on the AESIS (Agro-Environmental Sustainability Information Systems) framework, developed by Pacini et al. (2011). The project aims at identifying a list of indicators according to the different sustainability dimensions (environmental, economic and social) for the assessment of farms' sustainability performance and their contribution to the needs of the "Park System". Economic, biological and physical components describing the agro-ecosystem contribute to the overall sustainability (Belcher et al. 2004). Moreover, the complex trade-offs between these components claim for a holistic approach to agro-ecosystems sustainability assessment in order to identify sustainable management practices (Pacini et al., 2015). However, the dependency of farms activities on natural resources and human-made resources require a better understanding of the links between environmental indicators, farm management activities and policies. Integrated sustainability assessment tools may be appropriate to identify policies priorities for creating more sustainable agro-ecosystems.

2.4. Methodological framework

To account for the requirements of sustainability assessment of agro-ecosystems described above, we structured our analysis in three steps: (i) the life cycle environmental impacts assessment of the studied farms, (ii) the aggregation of some impacts categories into the ecosystem quality damage index, and (iii) the incorporation of this index into the SVA algorithm. Fig. 1 illustrates the approach to assess sustainability of agricultural production systems combining LCA and SVA.

Figure 1: A framework for an integrated sustainability assessment of Agro-ecosystems

[1] http://www.parcoaltamurgia.gov.it/officinadelpiano/index.php?option=com_content&view=article&id=856&catid=41



The sustainable value of different farms and agricultural sectors (specialized crop and mixed farms) is calculated to compare their role in guaranteeing the sustainability of agro-ecosystems. The farms' contribution to environmental sustainability can be monitored using LCA. Within the LCA methodological framework, the ReCiPe endpoint impact assessment method (Goedkoop et al., 2012) was selected in order to combine a problem (CML) and a damage oriented (Eco-indicator 99) approaches. Although traditional LCA is a steady-state tool which does not account for the uniqueness of the environmental systems affected and their sensitivities to emissions sources (Reap et al., 2008) this bias has been reduced by means of:

1. Consideration of only the most affected environmental impact categories by this site-specificity bias, such as: terrestrial acidification, freshwater eutrophication, soil and freshwater ecotoxicity (Reap et al., 2008).
2. Further reduction of the impact categories according to the main geo-morphological and ecological characteristics of the studied area

While the ReCiPe method uses the data on registered species at the European or Global level, in this study, the selected impact categories were normalized using data at the Mediterranean spatial level[2]. The ReCiPe methodology assumes that the quality of ecosystems is adequately represented by the diversity of species (Goedkoop et al., 2009). Hence, the five impact categories terrestrial acidification, terrestrial ecotoxicity and freshwater ecotoxicity, freshwater eutrophication and natural land transformation (measured in terms of *species lost*yr)* have been considered as good proxy for the damages caused to ecosystems quality. Assuming a linear relationship, an aggregated index has been designed (the ecosystem quality damage index), accounting for the overall effects of the farm's management activities on ecosystems quality. The ecosystem quality damage index has been incorporated into the SVA algorithm representing the natural resources used by farms to create value added for the society. However, by definition, the outcomes of the SVA compensate for the negative impacts generated by farms with the positive ones. Therefore,

[2] Data form 2000 have been used according to Brooks et al. (2002) in order to be consistent with the normalization procedure used into ReCiPe impact assessment method.

the value contribution (the Return to Cost ratio) for each category of capital was calculated in order to identify on which resource category (capital, land, labour, natural) the efforts should be focused in order to achieve a more sustainable agro-ecosystem within the Park.

3. - Materials and methods

To broaden the general insights on the integration and combination of sustainability assessment tools and to answer the call for methodological pluralism in holistic sustainability assessment (Gasparatos et al., 2009), this study performs a sustainability evaluation of farming systems both at the farm level and at the regional level. Therefore, LCA and SVA are integrated. Combining these two methods is feasible because they satisfy the request of complementarity, consistency and ability to address all the perspective of sustainability (Van Passel and Meul, 2012).
Application of this method is illustrated in a case study involving 14 mixed and specialized crops farms located in the Park. All the relevant farm characteristics are summarized in Table 1.

Table 1: Average descriptive statistics of the data sample of Crop and Mixed farms

	Unit	Crop Farms		Mixed Farms	
Farm size and Land use		Mean value	Range	Mean value	Range
Cultivated area (UAA)	ha	178	40 - 410	313	94 - 1040
Crops area	ha	178	40 - 410	60	4 - 121
Grassland area	ha			224	19 - 1000
Forage area	ha			40	9 - 67
Farm intensity					
Annual crop production	q./ha	20	3 - 37	26	15 - 56
Annual livestock production[a]	q./yr			56	0 - 150
Herd size	number of heads			293	90 - 520
Financial capital	KEUR	96	22 - 318	173	16 - 307
Subsidies	.KEUR	70	14 - 126	30	4 - 44
Labour	Average Work Unit	1	0,1 - 2	2	1 - 2

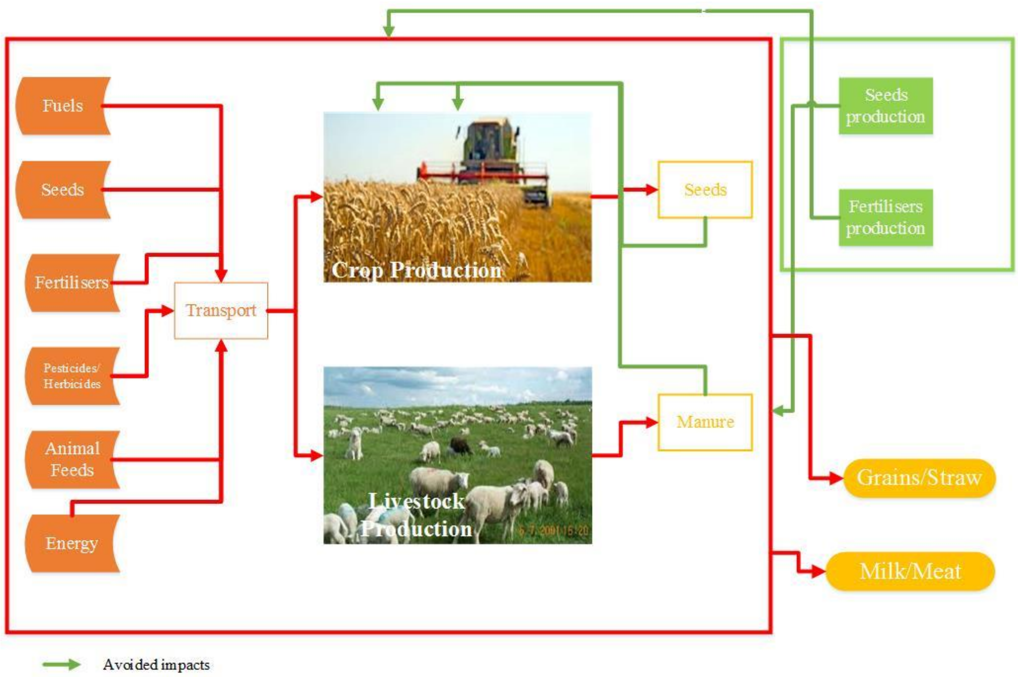
[a] The production of one of the mixed farms was excluded by the calculation because it is the only case that produce sheep meat

3.2 The LCA approach

LCA was applied for analysing the interactions between agricultural activities and the environment, allowing the evaluation of the main environmental impacts of farm activities in the Park area. The goal of this LCA study was to assess the relationships between farm activities and ecosystems quality loss within the Park. Data of the commercial farms that participated in the project "Agroecosistemi" (n=14) were used and refer to the year 2013. Data are collected on farm management strategies, yields, fertilizers and pesticide uses, water consumption, as well as the technique of animal husbandry (semi-wild or tethering), types and amount of animals feeding materials, etc. Data acquisition was performed using questionnaires that have been provided to participating farmers. An area based functional unit (FU) was defined for this study, since the sampled farms belong to the same class of "land use intensity" (*see Section 3.1*). In order to account for land size effect, each farm is considered as a single production unit, and it has been employed as reference for the estimation of environmental impacts.

The FU used within this study is thus a farm with UAA equal to 40 ha, which corresponds to the surface of the less extensive farm in our sample. For each farm, a detailed cradle-to-farm-gate life cycle assessment, including on and off farm pollution and avoided impacts, was performed (Figure 2).

Figure 2: System boundaries used for the environmental impact assessment of the sampled farms.



The Ecoinvent database (version 2.2) was consulted, for collecting the data concerning raw materials production and transports. Simapro 7.3.3 was used as a calculation platform. Transports inside the farm were excluded from the system boundaries. The use of manure and recycling of seeds were accounted in the system as prevented impacts due to the avoided production of, respectively, nitrogen and phosphorus fertilizers and commercial seeds. The amount of fertilizer produced was determined based on the mean N and P content of respectively bovine and sheep manure (Brentrup et al., 2000, Azeez et al., 2010). The emissions of N fertilizer and manure were calculated according to Brentrup et al. (2000), using different references to estimate the N-balance for the different crops (Ryden et al., 1984; Köpke and Nemecek, 2010, Garabet et al., 1998). The leaching fraction of applied P fertilizers was estimated according to Nest et al. (2014). Pesticide emissions were assessed using the PestLCI model (Dijkman et al., 2012). Methane emissions to air and N_2O emissions to water and soil from livestock breeding and grazing were assessed using the IPCC tier 2 approach (IPPC, 2006).

Table 2: Life Cycle Inventory of Crop and Mixed farms (yearly based)

	Crop farms							Mixed farms						
	Farm 1	Farm 2	Farm 3	Farm 4*	Farm 5	Farm 6*	Farm 7*	Farm 8	Farm 9**	Farm 10	Farm 11	Farm 12	Farm 13	Farm 14
Land size (ha of UAA)	75	165	250	400	145	40	240	90	121	130	101	155	146	195
Herd size (n° of heads)								520	262	90	160	384	410	
Inputs - agriculture														
Gasoil (l)	7,150	6,750	30,500	51,780	58,000	4,233	11,076	8,906	13,195	4,768	7,715	7,900	7,884	9,805
Lubrificant oil (l)	179	152	763	1,376	1,528	103	277	344	330	112	193	198	325	245
Seeds (q)		169	348	835	1,758	68	298	153	235	385	160	42	117	209
Fertilisers (q)				1,260										
Herbicide (l)				2		60								
Pesticide (l)						120								
Manure (q)					14,820			320		100		1,815	160	4,518
Inputs - livestock														
Animal feed (q.li)									106	20	62		12	806
Electricity (kWh)								3,508	2,425	300	1,496	3,508	2,588	4,139
Gasoil (l)									269	31	200	300		1,660
Lubrificant oil (l)									7	1	5	8		42
Emissions - agriculture														
Carbon dioxide (q)	210.92	184.06	324.03	5,933.02	746.28	12,043.53	233.46	102.51	330.58	96.78	239.64	258.29	170.86	119.51

Carbon monoxide (kg)	55.68	33.17	56.83	1,457.08	121.34	3,722.36	48.10	18.35	42.04	12.48	38.66	37.67	30.94	16.11
Methane (kg)	0.88	0.77	4.84	24.66	3.10	50.45	0.97	0.76	1.37	0.42	1.00	1.07	0.71	0.50
Sulfur dioxide (kg)	6.83	5.98	10.57	191.93	24.22	391.48	7.59	3.59	10.71	3.14	7.78	8.37	6.31	3.87
Nitrogen oxides (q)	2.79	2.72	4.80	78.16	10.86	198.72	3.52	1.38	4.67	1.32	3.50	3.67	2.51	1.70
Ammonia (q)	0.001	0.001	1.60	552.90	37.13	5.69	114.88	0.03	0.002	1.03	0.002	0.05	1.65	91.83
Nitrate (q)				534.38	124.48	12.02	111.04	12.16		3.81		0.18	6.64	514.25
Phosphorus (q)					672.43			21.95		6.86		14.28	21.95	222.29
Fenoxaprop-p-ethyl ester (q)				0.03										
Emissions - livestock														
Methane (kg)								24.53	19.66	18.23	19.63	23.44	18.23	27.77
Nitrous oxide (kg)								548.84	5.36	262.04	501.92	1,604.20	2,139.51	3,083.67

* : Are managed using conventional agricultural practices

**: Livestock uses only indoor rearing techniques.

For the life cycle impact assessment (LCIA) the endpoint ReCiPe method (Goedkoop et al., 2012) was used, which integrates the 'problem oriented approach' of CML-IA (Guinée et al., 2002) and the 'damage oriented approach' of Eco-indicator 99 (Goedkoop and Spriensma, 2001). Both these approaches have strengths and weaknesses related to: (i) the level of uncertainty and (ii) the interpretability of the results. The Recipe methods implements both strategies and has both midpoint (problem oriented) and endpoint (damage oriented) impact categories. The "Alta Murgia" is the main water resource for the entire Apulia Region (Canora et al., 2008). Beside it is highly important in terms of vascular plants and animals biodiversity (Perrino et al., 2006 and Cotecchia, 2010). , To account for these typical traits, the impact categories used for this study were water - use and land-use changes (Chapin III et al., 2000). The ReCiPe normalization factors are based on data at both the European and global level, whereas policy makers often are interested in using smaller regions as reference system (Sleeswijk et al., 2008). In this study, the selected impact categories were normalized based on the rate of yearly species lost for the Mediterranean basin in the year 2000 as explained by Brooks et al. (2002).

Taking into account the "*conceptual and data limitations"* existing for the inclusion of biodiversity and ecosystems quality into the LCA framework (Toumisto et al., 2012, Curran et al., 2011; Schmidt, 2008, see also *section 2*) the selected impact categories were considered as a good proxy for assessing the damages produced by farm activities to the quality of ecosystems, landscapes and wildlife habitats. The others impact categories associate with the human health and resources areas of protection (see Goedkoop et al., 2012) were excluded from the study. The assumption for this choice was that the Park Authority was more interested in understanding how agriculture activities affected biodiversity and ecosystems' quality at the local level, which can provide a more direct link to political goals (Sleeswijk et al., 2008). Land occupation (agricultural and urban) impact categories are usually estimated based onthe species richness ignoring human distortion (De Schryver et al., 2010). Therefore, these impact categories are also excluded from the study to avoid overestimated damages.

3.3 The Sustainable Value Approach (SVA).

The SVA methodology assumes that a firm contributes to sustainable development whenever it uses its resources more efficiently than other companies, reducing or unchanging the overall resource used (Van Passel and Meul, 2012). The methodological steps to calculate the sustainable value of a firm are:

(i) Define the aims of the analysis and determine the addressed stakeholders.

(2) Determination of the relevant resources with regard to sustainability performance of the firms or the economic sector.

(iii) Determine the benchmark values. The benchmark determines the costs of the resource that a firm (or economic sector) must exceed in order to produce sustainable value.

(iv) Comparison of the productivity level of a company resource with the corresponding benchmark while keeping the overall resource use constant. When the productivity of the company exceeds the opportunity cost, the company contributes to a sustainable use of the considered resource.

The opportunity cost of a resource form is the cost of the most valuable alternative and can be calculated as:

$$opportunity\ cost\ = value\ added_{benchmark}/resource\ _{benchmark} \qquad (1)$$

A firm generates sustainable value by using resources more efficiently than the benchmark. Accordingly, the *value spread* by the *company$_i$*. is calculated by subtracting the opportunity cost from the efficiency of resource use for the company (2).

$$value\ spread_i\ = value\ added_i/capital_i\ -\ value\ added_{benchmark}/resource_{benchmark} \qquad (2)$$

Therefore, the sustainable value of the *company$_i$ is* assessed by summing up the value contribution for every category of resource (3) that will be estimated by multiplying the *value spread $_i$* for a certain category of resource by the amount of resource used by *company$_i$*.

$$sustainable\ value_i\ = \frac{1}{n}\sum_{s=1}\left(value\ spread_i^s * capital_i^s\right) \quad \begin{matrix} for\ s\ [1,k] \\ k = n°\ of\ resource \end{matrix} \qquad (3)$$

According to Van Passel et al. (2007), dividing by the number of resources *n* allows to correct for the overestimation of value created, avoiding double counting (Figge and Hahn, 2005).

In order to account for the company size, the Return to Cost Ratio (RTC) for farm *i* was calculated (Van Passel et al., 2009) according to equation 4.

$$Return\ to\ Cost\ ratio_i\ =\ value\ added_i/(value\ added_i - sustainable\ value_i) \qquad (4)$$

A RTC above one means that the company is more efficient in resource allocation than the benchmark. The most criticized aspect of this method is the definition of the benchmark (Mondelaers et al., 2011). This is due to the fact that that the method is not able to capture whether the overall resource use ensures a sustainable outcome (Figge and Hahn, 2004a); and so the benchmark may be defined in such a way that it does not describe a sustainable resource use (Ang et al., 2011). Although, the choice of the benchmark strongly affects the explanatory power of the analysis (Figge and Hahn, 2005), Van Passel et al. (2007) showed in an application on Flemish dairy farms that the ranking of the companies does not differ between several types of benchmarks. An interesting alternative approach is the construction of a sustainability benchmark using appropriate agro-environmental farm models (Merante et al., 2015). Unfortunately, these models were not available for the assessment of agricultural systems in the studied area.

For the above mentioned reasons, the average for each resource has been used as a benchmark. To test the robustness of the sustainable value calculations, the rank correlation (Spearman’s rho) of RTC using different benchmarks is calculated (Table 3). The correlations are high and significant.

Table 3: Correlation between the Return-to-cost ratio using different benchmarks

Return-to-cost	Benchmark 1	Benchmark 2	Benchmark 3
Benchmark 1[a]	1	0.9428***	0.6131**
Benchmark 2		1	0.6440**
Benchmark 3			1

[a] Benchmark base using the average for each form of resources

* significant at 10%   ** significant at 5%   ***significant at 1%

The different forms of capital considered were: (i) labour, (ii) farm capital, (iii) used land (ha), (iv) ecosystem quality damage (species lost*yr). For each farm, labour was measured in Annual Working Unit (AWU). Farm capital (assets) was calculated as the total capital minus the value of land to avoid double counting; while the ecosystems quality damage index was calculated by summing the considered environmental impact indicators of the LCA analysis. Therefore, in this study the Sustainable Value was expressed as a function of farm capital, used land, labour and ecosystems quality damage (Equation 5).

$$sustainable\ value_{\,i} = f\,(farm\ capital_{\,i}, used\ land_{\,i}, labor_{\,i}, ecosystem\ quality\ damage_{\,i}) \quad (5)$$

This highly relevant selection of several resources categories is ignored by previous studies (Van Passel et al. 2007; Van Passel et al. 2009). This is especially critical for natural resources for which the choice was merely data driven without a sound selection method (see Ang et al., 2011; Van Passel et al., 2009; Van Passel et al., 2007). Only Merante et al. (2015) and Pacini et al. (2015) used agro-environmental models to outline environmental thresholds that can be used as farm sustainability benchmarks.

4. - Results

There is a large within-group variability for the indicators scores between specialized crops farms and mixed farms. The ecosystem quality damage scores for the sampled farms range between 3.60E-05 and 3.89E-02 species lost*yr as shown in Table 4. Specialized crop farms have less impact on the environment in terms of cumulative ecosystems quality damages, accounting for almost the 30% of the total estimated damages to ecosystems (Table 4).

Table 4: Characterization of the environmental impacts of Crop and Mixed farms (species lost*yr-)

		Terrestrial acidification	Freshwater eutrophication	Terrestrial ecotoxicity	Freshwater ecotoxicity	Natural land transformation	Ecosystem quality damage index
Farm 1	CF	1.88E-06	2.84E-07	-3.89E-08	4.43E-08	3.38E-05	3.60E-05
Farm 2	CF	1.44E-05	3.47E-07	6.17E-06	1.79E-07	6.43E-05	8.54E-05
Farm 3	CF	-6.73E-06	4.72E-07	-1.48E-05	-1.87E-08	1.75E-04	1.54E-04
Farm 4	CF	4.01E-03	3.99E-05	1.21E-04	5.31E-06	3.42E-03	7.60E-03
Farm 5	CF	1.80E-04	2.14E-03	1.11E-02	1.38E-03	2.00E-04	1.50E-02
Farm 6	CF	9.64E-05	2.27E-05	2.75E-05	2.46E-06	3.53E-04	5.02E-04
Farm 7	CF	1.03E-03	8.34E-06	1.98E-05	1.01E-06	4.82E-04	1.54E-03
Farm 1	MF	-6.94E-05	2.82E-03	1.46E-02	1.82E-03	8.38E-05	1.92E-02
Farm 2	MF	6.62E-06	2.21E-06	1.94E-05	3.76E-07	3.22E-04	3.51E-04
Farm 3	MF	7.54E-05	7.46E-04	3.86E-03	4.80E-04	7.86E-05	5.24E-03
Farm 4	MF	-3.79E-06	1.32E-08	1.36E-07	-4.28E-08	4.97E-05	4.60E-05
Farm 5	MF	2.39E-05	7.92E-04	4.09E-03	5.09E-04	5.38E-04	5.95E-03
Farm 6	MF	-3.93E-05	1.91E-04	9.72E-04	1.23E-04	1.26E-04	1.37E-03
Farm 7	MF	6.25E-04	5.58E-03	2.89E-02	3.59E-03	1.40E-04	3.89E-02
SD		0.001	0.002	0.008	0.001	0.001	0.01

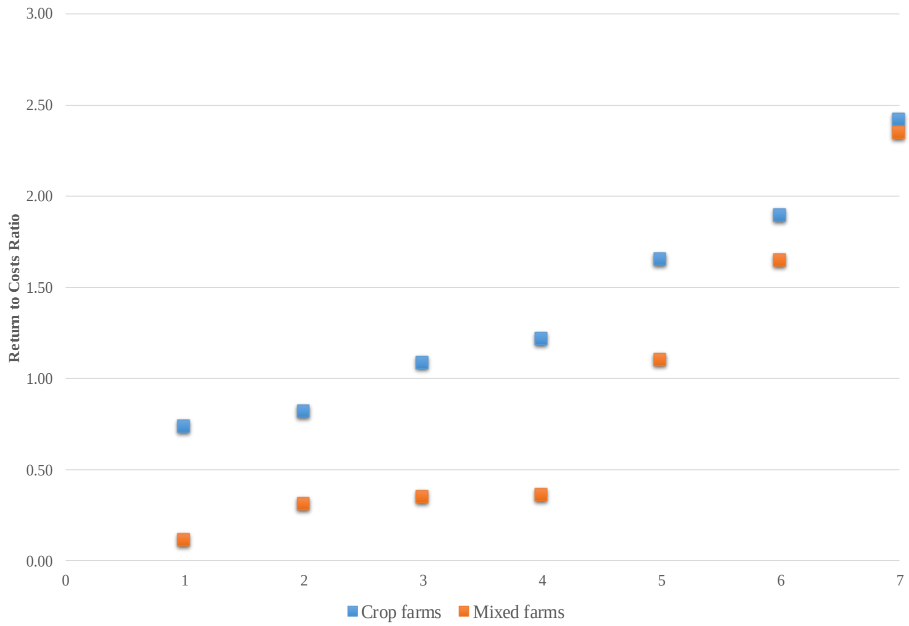
CF = crop farms; MF = mixed farms; SD = standard deviation

Specialized crop farms score better for freshwater use and terrestrial ecotoxicity; while they have higher impacts for terrestrial acidification and transformation of natural land. Farm 5 CF and Farm 1 MF show significant impacts in terms of terrestrial ecotoxicity which consistently affect the overall ecosystem quality damage outcome. These high impacts are due to the consistent amount of manure used (Table 2), which consequently determines a high level of phosphorous leached into water bodies.

These results can be explained by the higher use of human-made resources for crop farms such as gasoil, seeds, fertilizers and pesticides. Usually, mixed farms produce only the forage needed for feeding the livestock and use natural pastures for grazing their animals. Therefore, they have less cultivated land for crop production, leading to a decreasing number of soil tillage operations and a less intensive use of chemicals. Moreover, seed' recycling is more widely practiced in mixed farms generating lower impacts on soil, natural land transformation and climate changes. The higher impacts of mixed farms on freshwater (ecotoxicity and eutrophication) and terrestrial ecotoxicity are determined by freshwater nitrogen and phosphorus leaching as a result of animals grazing and manure management.

The performance of the crop and mixed farms clearly differs (Figure 3). Overall, most of the specialized crop farms are sustainable showing a RTC above 1, whereas most of the mixed farms are less sustainable showing RTC below 1. However, both farm groups exhibit frontrunners with a RTC above 1.

Figure 3: Return to cost ratio using the average benchmarks



The variables in our data set that may explain the difference in farms performance are the capital productivity and eco-efficiency (Table 5).

Generally, the most sustainable farms maximize the productivity of capital, labour and land while minimizing the ecosystem quality damage index. Mixed farms perform well in terms of land productivity, while specialized crop farms achieve better results in terms of labour and capital productivity and have a lower impact on ecosystem quality. From these calculations of the sustainable value, it can be concluded that the focus should be put on the reduction of ecosystem quality damages of mixed farms. Further, Higher land productivity of crop farms are important to strengthen the sustainability performance of agricultural activities within the Park.

Table 5: Average resources productivities and eco-efficiency of Crop and Mixed farms.

	Capital productivity (€/€)	Labour productivity (M€/AWU)	Land productivity (€/ha)	Eco-efficiency (€/species lost *yr)
Crop farms	1.79	1.40	514.09	2.82E+08
Mixed farms	0.997	0.44	848.90	1.75E+08

5. – Discussions and Conclusions

In this paper, we explored the possibilities to integrate biophysical and monetary sustainability assessment tools through combining the impacts of agriculture activities on ecosystems with the concept of natural capital. To achieve this goal, we performed a case study where Life Cycle Assessment and Sustainable Value Approach were simultaneously used to assess the sustainability of agricultural systems within the Park. The methodology presented in this study allowed an integrated assessment of the economic and environmental dimensions of sustainability, providing decision-makers an overview of the effects of agriculture activities on local sustainable development. Moreover, the use of a benchmark to measure the overall performance of farms and their relative efficiency can be useful to highlight opportunities of improvement both at farm and regional level. The main goal was to develop a novel framework for combining biophysical and monetary oriented tools to assess sustainability of agricultural systems. However, considering the large variability in farm accountancy data and agriculture management practices, a higher number of farms needs to be sampled, in order to avoid the inference on outcomes of frontrunners and laggards. Further research is needed to improve the benchmarks such as the efficiency frontiers which require more data availability in order to guarantee the robustness. Although further improvement is needed, the new methodology for measuring farm sustainability proofed to be promising.

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