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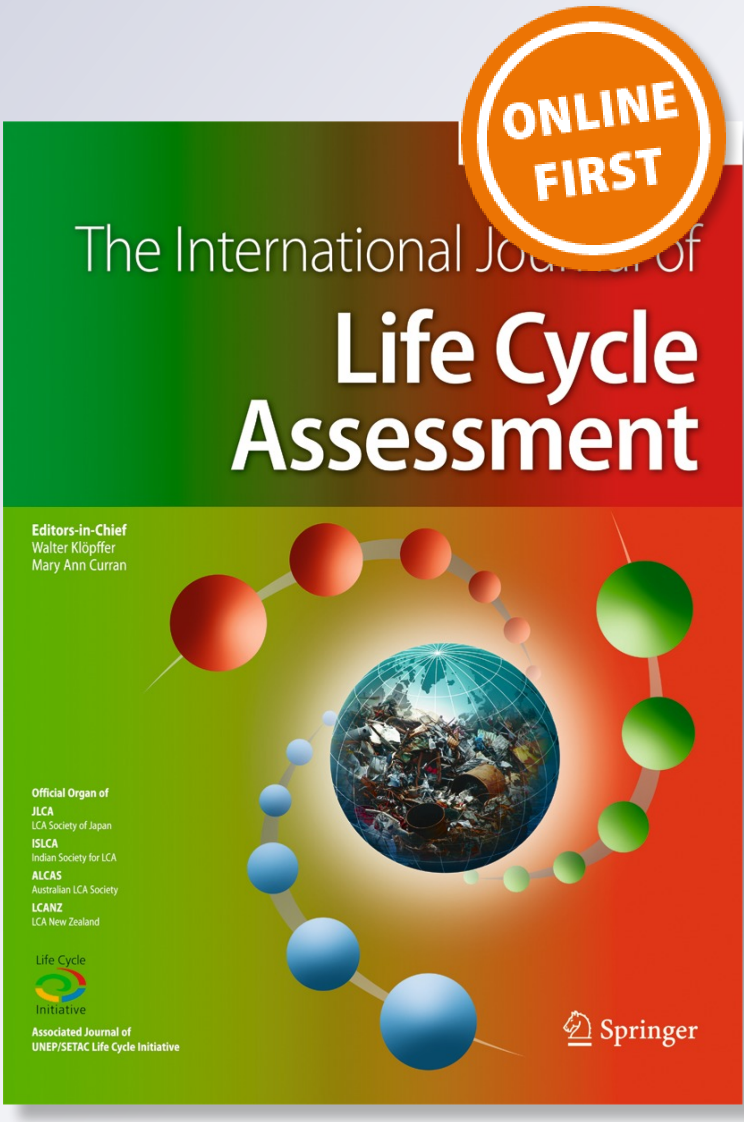

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LAND USE IMPACTS

Assessing potential desertification environmental impact in life cycle assessment. Part 2: agricultural case study in Spain and Argentina

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Abstract

Purpose Land use in dry lands can result in a final stage where land is completely depleted or entirely degraded causing the desertification phenomenon. The first part (part 1) of this series of two articles proposed a methodology to include desertification in life cycle assessment (LCA). A set of variables to be measured in the life cycle inventory, characterization factors, and an impact assessment method for the life cycle impact assessment phase were proposed. This second part (part 2) aims at showing the application of the model proposed in part 1 on two case studies of agricultural activities.

Methods The impact model proposed is applied to plots of land devoted to agricultural activities in two countries: Argentina and Spain. In the agricultural plots of Spain (1SP to 9SP), two crops were analyzed: winter wheat (*Triticum aestivum*) and rapeseed (RS, *Brassica napus*). Two crops were considered in the Argentinean case study: rapeseed (RS, *B. napus*) and digit grass (*Digitaria eriantha*) (10AR to 17AR). A bare soil state is considered in both countries as a reference state. Both case studies consider only the agricultural stage in the inventory of a complete life cycle assessment study. Both also consider only one impact category in life cycle environmental assessment: desertification impact due to land occupation.

Results and discussion On the basis of the obtained results, it can be inferred that cultivating 1 ha of rapeseed and 1 ha of wheat has the same impact on the analyzed plots in Spain and improves the reference state conditions in 50 % of the cases. Moreover, rapeseed grown in Mendoza produces almost the same impact as in some of the Spanish plots. Normalized areas of plots could be useful to compare results in different regions of the world to avoid the influence of the area of occupation in results.

Conclusions The proposed model implies a contribution of significant importance because so far there has not been an impact assessment tool for land use in dry lands within the LCA framework. The main strength of the proposed model is that it allows a simple way to quantify the desertification impact. Also, it is emphasized that the model can be adapted virtually without difficulty to the evaluation of all types of crops with different management practices in different regions in the life cycle impact assessment stage.

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1 Introduction

The environmental impact of land use and land use change as a consequence of intensive human activities has gradually gained practical recognition in life cycle assessment (LCA) (Koellner et al. 2013a, b), a method that aims at evaluating the environmental impacts of goods and services. In the last decade, many new methods to address land use impacts on biodiversity and ecosystem services have been developed (Koellner and Scholz 2008; Maia de Souza 2010; Núñez et al. 2013), allowing the inclusion of these relevant aspects in LCA. But all of them did not consider explicitly land use in dry lands.

Land use in dry lands can result in a final stage where land is completely depleted or entirely degraded. The consequences of the use of land in dry lands differ from those of other climatic areas because of the inability of already degraded lands to be recovered and, consequently, the expansion of the desert area. Desertification leads to loss of soil quantity and quality and biodiversity. In 1992, during the Rio Conference, desertification was defined as *degradation of arid, semiarid, and dry–subhumid lands as a consequence of many factors including climatic variations and human activities*. According to the first article of the United Nations Convention to Combat Desertification (United Nations 1994), dry lands are divided by their relation between precipitation and potential evapotranspiration (P/ETP) resulting in: (a) hyper-arid lands, P/ETP <0.05; (b) arid lands, P/PET between 0.05 and 0.20; (c) semiarid lands, P/PET between 0.20 and 0.50; and (d) dry–subhumid lands, P/PET between 0.50 and 0.65.

According to the United Nations, "41 % of the world's land is occupied by dry lands where 2 billion people live. Between 10 % and 20 % of these lands—more than 4,000 million hectares—are degraded or unproductive" (MA 2005). Not only the use of land is critical in dry lands, but also it must be considered that dry lands suffer from water stress and competition for scarce water resources. Environmental impacts of water use and consumption in dry lands can be evaluated in LCA combining the long-standing emission-oriented impact categories (i.e., eutrophication, acidification, and toxicity) with the recent categories for water consumption (Kounina et al. 2012).

These assertions revealed the importance of including land use in dry lands as a special type of land use impact within this impact category in LCA. With this aim, the first part of this series of two articles (Núñez et al. 2010, referred to as "part 1" in the article), suggested a methodology to assess desertification within the LCA framework. A set of variables to be measured in the life cycle inventory (LCI), characterization factors (CFs), and an impact assessment method for the life cycle impact assessment (LCIA) phase were proposed. The multi-indicator model requires, in the LCI stage, the estimation of four environmental variables linked to the activity evaluated: aridity index, erosion, aquifer overexploitation, and fire risk. The quantitative figures estimated are transformed into semiquantitative values, as shown in Table 1. More details about the procedure and data range values of this table can be found in part 1. The four indicators of the LCI were selected from an extended list of desertification indicators agreed by many researchers and decision makers from the scientific community, taking into account their applicability and data availability in the LCA context.

This second article shows the application of the model proposed on two case studies. The purpose is, firstly, to illustrate the method developed with an example of agricultural land use. Secondly, it aims at verifying the applicability and feasibility of the model identifying its strengths and weaknesses. Moreover, this second article is intended to demonstrate the existing need of further researching and developing accurate life cycle impact assessment methods to assess land use in dry lands.

2 Methods

With the aim of achieving the mentioned objectives, the LCA desertification model developed in part 1 is applied to plots of land devoted to agricultural activities in two countries (Argentina and Spain). The model consists in registering information on four biophysical variables related to the desertification phenomenon (aridity, soil erosion, aquifer overexploitation, and fire risk, with a created scale of values); the geographical location of the activity; and finally, the spatial and temporal extension of the activity. CFs to establish desertification risk were measured for the main terrestrial natural regions (ecoregions) by means of geographic information system (GIS) and vary between 4.0 in the ecoregion with the lowest desertification risk and 7.6 in the ecoregion with the highest desertification risk.

Further details about the agricultural systems studied are explained in the following sections.

2.1 Agricultural systems studied in Spain

In the agricultural plots of Spain, two crops were analyzed: winter wheat (*Triticum aestivum*) and rapeseed (RS, *Brassica napus*). The cereal (wheat) is one of the most broadly cultivated food crops in the country. Rapeseed is an herbaceous crop that has gained a large cultivation area to produce biodiesel during recent years. Oilseed would be a suitable energy crop for growing in Spain as an alternative to cereal for increasing the share of renewable energy consumption within the country. Both crops are winter annual

Table 1 Multi-indicator system of the inventory phase ($LCI_{Desertification}$, dimensionless)

Desertification variables	
Estimation value	Evaluation (LCI variable data, dimensionless)
Aridity variable ($V_{Aridity}$)[a]	
Arid (0.05–0.20)	3
Semiarid (0.20–0.50)	2
Dry–subhumid (0.50–0.65)	1
Humid–subhumid (0.65–0.75)	0
Humid (>0.75)	0
Erosion variable ($V_{Erosion}$)	
>25 t ha^{-1} $year^{-1}$	3
12–25 t ha^{-1} $year^{-1}$	2
<12 t ha^{-1} $year^{-1}$	1
Aquifer overexploitation variable ($V_{Aquifer\ overexploitation}$)	
$W>0.8R$	2
$0.8R\geq W>0.4R$	1.6
$0.4R\geq W>0.2R$	1.3
$W\leq 0.2R$	1
Fire risk variable ($V_{Fire\ risk}$)	
≥10 % burned area in previous 10 years	2
<10 % burned area in previous 10 years	1

W withdrawal, *R* recharge

[a] Ratio between precipitation (*P*) and evapotranspiration (ETP) (*P*/ETP), dimensionless

crops, with the seeding date between October and December and the harvesting date between June and July, depending on the climatic conditions of the region where they grow. Farmers usually grow wheat and rapeseed in rotation systems of two or more years, completing the other rotation years with legumes and other cereals (barley, oats) or keeping the soil unproductively fallow.

Nine agricultural plots were assessed in Spain, six were located in the region of Catalonia (northeast of Spain) and three located in the region of Soria (center of Spain). In Catalonia, the plots were in two different districts: the Empordà District (plot codes 1SP, 2SP, 3SP) and the Noguera District (plot codes 4SP, 5SP, 6SP). Each of the districts has its own climate. In the Noguera District (west of Catalonia), rainfalls are lower and temperatures are higher than in the Empordà District (northeast of Catalonia) (Ninyerola et al. 2001). More than 55 % of the herbaceous crops in the Empordà District grow in rainfed conditions, while the percentage rises to nearly 65 % in the Noguera District (DARP 2010). Plots in the Soria region were located in three districts: Tierras del Burgo (plot code 7SP), Soria (plot code 8SP), and El Moncayo (plot code 9SP). They share climatic parameters: annual rainfall and temperature are similar to that of the Noguera District in Catalonia. In Soria, many more rainfed crops are cultivated than irrigated crops (Junta de Castilla y León 2009a). The two analyzed crops, winter wheat and rapeseed, are usually grown in rainfed conditions in the two areas, as well as in the rest of Spain. Due to their climatic conditions, both Catalonia and Soria regions seem to be prone to desertification.

2.2 Agricultural systems studied in Argentina

Two different crops were considered in the Argentinean case study: rapeseed (RS, *B. napus*) and digit grass (*Digitaria eriantha*). Unlike what happens in Spain, in Argentina, the development of energy crops, except soybeans, have not been extended yet. There are many initiatives to mainstream alternative energy crops to soybean or sugar cane all around the country. However, so far all, are in the experimentation stage to evaluate the crop adaptation to climate, water requirements, productivity, and other concerning issues. Rapeseed is a crop recently introduced into the agricultural circuit of the western region of Argentina, where plots of rapeseed were located. The potential of rapeseed in this area seems to be significant because it is a winter/spring crop that does not compete with vines (the main agricultural activity of Mendoza) for the use of water. This is the reason why rapeseed can work very well in these dry irrigated lands (Silva Colomer 2008). On the other hand, digit grass is a very common forage species that grows really well in the semiarid central region of Argentina. This is a summer-growing perennial grass, possessing better quality protein than other grasses (Veneciano et al. 2006), and is a very persistent, productive, and draught-tolerant species (Buckley 1959).

The case study for Argentina consisted of eight plots, five located in the Junín District (east of Mendoza city, in the west of Argentine, plot codes 10AR, 11AR, 12AR, 13AR, 14AR) and three in rainfed conditions in the Gobernador Dupuy District, south of San Luis City (plot codes 15AR, 16AR, 17AR). In Mendoza, there are no crops in rainfed conditions because annual precipitations cannot supply enough water to grow any possible crop, having the mean precipitation rate of 188 mm/year (DAWC 2011).

2.3 General considerations

Apart from the study of the desertification impact due to the cultivation of wheat and rapeseed in Spain and digit grass and rapeseed in Argentina, a third common reference situation was included in the assessment: unproductive soil in fallow land. Fallowing is used between two crop periods to promote soil nutrient and moisture recovery for growing subsequent crops while reducing the dependence on external inputs. The unseeded fallow land management system is the most common in arid and semiarid zones such as those of the studied plots, where soil–water is the main limiting factor and a seeded fallow land would refrain from moisture recovery.

Both case studies consider only the agricultural stage in the inventory of a complete life cycle assessment study of, for example, 1 MJ from soybean or rapeseed biodiesel or one slide of 30 g of wheat bread. Both also consider only one impact category in life cycle environmental assessment: desertification impact due to land occupation but not for land transformation. This portion of a LCA could be integrated into a complete LCA if it is needed.

The evaluation of the desertification impact of growing energy crops will provide a new insight into the environmental local impacts due to energy crop cultivation; the environmental sustainability of which is still under debate. Despite being a common forage species, digit grass is not being fully incorporated into the agricultural circuit of San Luis. That is why, by considering the digit grass case study, the possible impact that could result from its permanent incorporation into the agricultural circuit of San Luis can be assessed. Finally, with the study of a traditional local crop in Spain (winter wheat) and the common reference of soil in fallow land in both countries, desertification impact of traditional crops and agricultural practices can be compared to impacts of crops spread over the last years dedicated to energy end uses.

3 Results

3.1 Inventory

The desertification model suggested in part 1 defined a set of variables to be measured in the LCI: (1) the geographical location of the activity; (2) the spatial and temporal extension of the activity; and (3) the four biophysical variables of aridity index (V_{Aridity}), erosion (V_{Erosion}), aquifer overexploitation ($V_{\mathrm{Aquifer\ overexploitation}}$), and fire risk ($V_{\mathrm{Fire\ risk}}$). The procedure followed to obtain the value of each individual variable is described in the next sections. Tables 2, 3, 4, 5, 6, 7, and 8 summarize these values as well as the final figure for the inventory phase ($\mathrm{LCI}_{\mathrm{Desertification}}$).

3.1.1 Geographical location

Recording the geographical location of the plot (see Table 2) can assist in identifying the environmental conditions at the given location if the LCA practitioner uses GIS to gather the LCI data. The use of GIS has spread over the last decade and it facilitates greatly obtaining and processing spatial data. For example, precipitation for V_{Aridity} and groundwater withdrawal for $V_{\mathrm{Aquifer\ overexploitation}}$ might be obtained from GIS data sets. On the other hand, the geographical location of the plot must be available for the LCIA phase, as CFs vary spatially. CFs were established for the main 15 biogeographical regions of the world, following the Bailey's ecoregion classification (Bailey 2002, see part 1). Therefore, to assign each plot to the right ecoregion and CF, the location of the plot should be known. It is recommended to report the location as exactly as possible (longitude/latitude whenever possible). Failing that, a broader resolution (e.g., region) can be used, although this will both reduce the quality of the LCI data if these are derived using GIS data sets and can lead to a wrong selection of the ecoregion where the activity takes place in the LCIA phase.

3.1.2 Spatial and temporal extension of the activity

All the studied crops are evaluated over the period of 1 year, from sowing until the harvest of the crop and a residue cover plus a rough fallow period to complete the overall year. Doing so, the temporal variable is normalized in all the plots. To include the spatial variable, some consideration has to be made because huge differences were found among the plots assessed. Argentinean plots differ at least in two orders of magnitude from the Spanish ones. Thus, with the purpose of having a uniform spatial unit of comparison, 10,000 m^2 (1 ha) is considered in all plots. The spatial (in square meter) and temporal (years) extension of the occupation is shown in Table 2 for each plot.

3.1.3 Aridity variable

Aridity is defined as the ratio between precipitations (*P*) to potential evapotranspiration (ETP; United Nations, 1994). Both meteorological parameters were obtained from local available public statistics of the closest weather station to

Table 2 Identification code, location and time and area of occupation of the assessed plots

District (*comarca/departamento*) of the case study	Plot code	Geographic coordinates (X, Y)[a]	Occupation area (m^2)	Occupation time (years)
Empordà	1SP	(3.1776E, 42.0117N)	10,000	1
Empordà	2SP	(3.0608E, 42.1730N)	10,000	1
Empordà	3SP	(3.0430E, 42.2028N)	10,000	1
Noguera	4SP	(0.9233E, 41.6777N)	10,000	1
Noguera	5SP	(0.8107E, 41.7218N)	10,000	1
Noguera	6SP	(0.8097E, 41.7500N)	10,000	1
El Moncayo	7SP	(1.9786W, 41.7680N)	10,000	1
Tierras del Burgo	8SP	(3.2204W, 41.5650N)	10,000	1
Soria	9SP	(2.4289W, 41.6090N)	10,000	1
Junín	10AR	(68.4844W, 33.1158S)	10,000	1
Junín	11AR	(68.4844W, 33.1158S)	10,000	1
Junín	12AR	(68.4844W, 33.1158S)	10,000	1
Junín	13AR	(68.4844W, 33.1158S)	10,000	1
Junín	14AR	(68.4844W, 33.1158S)	10,000	1
Gobernador Dupuy	15AR	(66.2026W, 34.7255S)	10,000	1
Gobernador Dupuy	16AR	(66.1903W, 34.7240S)	10,000	1
Gobernador Dupuy	17AR	(66.1776W, 34.7227S)	10,000	1

[a] Longitude and latitude with WGS84 datum

each plot. As recommended by FAO, the Penman–Monteith method was used to determine ETP (Allen et al. 1998). Data for 1SP to 6SP plots were obtained from the Catalan regional government (RuralCat 2009) for the period 1990–2008, and for 7SP, 8SP, and 9SP plots, data were from MARM (2009a), for which only complete statistics for the period 2005–2008 could be gathered. The information used in the Argentinean plots came from the Department of Agriculture and Weather Contingency. Statistics are for the period 1999–2009 in AR10 to AR14 and for the period 2002–2010 in 15AR to 17AR plots (DAWC 2011; personal conversation with Carlos Frasinelli, from INTA, Villa

Table 3 Precipitation, potential evapotranspiration, aridity index, and $V_{Aridity}$ of each assessed plot

Plot code	Precipitation (P, Lm^{-2} $year^{-1}$)	Potential evapotranspiration (ETP, Lm^{-2} $year^{-1}$)	Aridity index (P/ETP, dimensionless)	Climate type	$V_{Aridity}$
1SP	621.0	907.0	0.68	Humid–subhumid	0
2SP	621.0	907.0	0.68	Humid–subhumid	0
3SP	621.0	907.0	0.68	Humid–subhumid	0
4SP	348.1	964.9	0.36	Semiarid	2
5SP	348.1	964.9	0.36	Semiarid	2
6SP	348.1	964.9	0.36	Semiarid	2
7SP	377.3	1,187.6	0.32	Semiarid	2
8SP	421.9	1,084.8	0.39	Semiarid	2
9SP	395.1	1,073.0	0.37	Semiarid	2
10AR	187.9	1,391.3	0.14	Arid	3
11AR	187.9	1,391.3	0.14	Arid	3
12AR	187.9	1,391.3	0.14	Arid	3
13AR	187.9	1,391.3	0.14	Arid	3
14AR	187.9	1,391.3	0.14	Arid	3
15AR	986.1	1,864.0	0.50	Semiarid	2
16AR	986.1	1,864.0	0.50	Semiarid	2
17AR	986.1	1,864.0	0.50	Semiarid	2

Table 4 Values of the USLE factors for each considered plot

Plot code	R	K	LS	C, WW	C, RS	C, D	C, fallow	P
1SP	1,634.2	0.0276	0.3411	0.300	0.237	n.a.	0.9	1
2SP	1,930.0	0.0465	0.2200	0.339	0.243	n.a.	0.9	1
3SP	2,277.5	0.0331	0.6545	0.346	0.261	n.a.	0.9	1
4SP	1,244.5	0.047	3.2657	0.323	0.266	n.a.	0.9	1
5SP	917.9	0.040	0.8228	0.310	0.260	n.a.	0.9	1
6SP	1,244.5	0.055	2.4763	0.323	0.266	n.a.	0.9	1
7SP	844.7	0.0514	0.5703	0.269	0.239	n.a.	0.9	1
8SP	536.8	0.0529	0.1019	0.258	0.211	n.a.	0.9	1
9SP	580.2	0.0442	0.9914	0.269	0.219	n.a.	0.9	1
10AR	467	0.0085	0.1276	n.a.	0.535	n.a.	0.9	1
11AR	467	0.0520	0.1276	n.a.	0.530	n.a.	0.9	1
12AR	467	0.0537	0.1276	n.a.	0.528	n.a.	0.9	1
13AR	467	0.0404	0.1276	n.a.	0.526	n.a.	0.9	1
14AR	467	0.0414	0.1276	n.a.	0.528	n.a.	0.9	1
15AR	–[a]							
16AR								
17AR								

Fallow is a reference

WW winter wheat, *RS* rapeseed, *D* digit grass, *n.a.* not assessed

[a]USLE information was given in a personal communication by Dr. Carlos Frasinelli from INTA EEA Villa Mercedes, San Luis, Argentina

Mercedes, San Luis, 2011 and 2012). According to data, aridity index of plots 1SP, 2SP, and 3SP corresponds to a humid–subhumid climate, of plots 4SP to 9SP and 15AR to 17AR to a semiarid climate, and of plots 10AR to14AR to an arid climate. The corresponding values of V_{Aridity} in the LCI phase for each plot are collected in Table 3.

3.1.4 Erosion variable

V_{Erosion} reflects water erosion of the activity under study. Water erosion is one of the main reasons for soil degradation and desertification, hence the importance to include it in the assessment. The universal soil loss equation (USLE

Table 5 Erosion and V_{Erosion} of each assessed plot

Plot code	WW erosion (t ha^{-1} year^{-1})	RS erosion (t ha^{-1} year^{-1})	D erosion (t ha^{-1} year^{-1})	Fallow erosion (t ha^{-1} year^{-1})	WW–V_{Erosion}	RS–V_{Erosion}	D–V_{Erosion}	Fallow–V_{Erosion}
1SP	4.6	3.6	n.a.	13.8	1	1	n.a.	2
2SP	6.7	4.8	n.a.	17.8	1	1	n.a.	2
3SP	17.1	12.9	n.a.	44.4	2	2	n.a.	3
4SP	61.6	50.8	n.a.	171.9	3	3	n.a.	3
5SP	9.4	7.8	n.a.	27.2	1	1	n.a.	3
6SP	54.7	45.1	n.a.	152.5	3	3	n.a.	3
7SP	6.7	5.9	n.a.	22.3	1	1	n.a.	2
8SP	0.7	0.6	n.a.	2.6	1	1	n.a.	1
9SP	6.8	5.6	n.a.	22.9	1	1	n.a.	2
10AR	n.a.	0.4	n.a.	0.6	n.a.	1	n.a.	1
11AR	n.a.	1.8	n.a.	3.0	n.a.	1	n.a.	1
12AR	n.a.	1.8	n.a.	3.2	n.a.	1	n.a.	1
13AR	n.a.	1.4	n.a.	2.3	n.a.	1	n.a.	1
14AR	n.a.	1.4	n.a.	2.4	n.a.	1	n.a	1
15AR	n.a.	n.a.	0.3	0.3	n.a.	n.a	1	1
16AR	n.a.	n.a.	0.3	0.4	n.a.	n.a	1	1
17AR	n.a	n.a.	0.3	0.4	n.a.	n.a.	1	1

Fallow is a reference

WW winter wheat, *RS* rapeseed, *D* digit grass, *n.a.* not assessed

Table 6 Groundwater balance and $V_{\text{Aquifer overexploitation}}$ values of aquifers for each assessed plot

Plot code	Withdrawal (W, hm^3 $year^{-1}$)	Recharge (R, hm^3 $year^{-1}$)	Water exploitation ratio (W/R)[a]	$V_{\text{Aquifer overexploitation}}$
1SP	25.2	73.2	0.3	1.3
2SP	22.0	48.1	0.5	1.6
3SP	22.0	48.1	0.5	1.6
4SP	72.6	95.1	0.8	1.6
5SP	72.6	95.1	0.8	1.6
6SP	2.0	33.5	0.1	1
7SP			0.1[a]	1
8SP	0.22 mm $year^{-1b}$		0.1[a]	1
9SP			0.1[a]	1
10AR	622	590	1.05	2
11AR	622	590	1.05	2
12AR	622	590	1.05	2
13AR	622	590	1.05	2
14AR	622	590	1.05	2
15AR	0	–	–	1
16AR	0	–	–	1
17AR	0	–	–	1

[a]Water exploitation ratio directly reported in CHDuero (2005)

[b]Maximum sustainable extraction (limestone area): 15 mm $year^{-1}$ (CHDuero 2005)

equation, Eq. 1; Wischmeier and Smith 1978) was the model selected in part 1 to estimate soil losses. According to USLE, soil loss is calculated by Eq. (1). Results for each considered plot are shown in Table 4.

$$A = R \times K \times LS \times C \times P \tag{1}$$

where A is the average soil erosion rate (in tons per hectare per year), relying on a rainfall pattern (R factor—rainfall and runoff factor), a specific soil type (K factor—soil erodibility factor), a topography (LS factor—topographic factor), and a possible combination of cropping system (C factor—cover and management factor) and conservation practices (P factor—support practice factor). Each USLE factor was calculated as indicated below:

R factor The R factor (MJ/ha.año) (mm/h) for the Spanish plots (1SP to 9SP) came from statistics of MARM (2009b), which has public detailed local rainfall datasets recorded for

Table 7 Percentage of surface affected by forest fires during the latest 10 years and $V_{\text{Fire risk}}$ in the regions where plots were located

Plot code	District area (10^4 m^2)	Forest fires area (10^4 m^2)	% of affected area	$V_{\text{Fire risk}}$
1SP	70,170	2,454	3.5	1
2SP	70,170	2,454	3.5	1
3SP	135,754	9,150	6.7	1
4SP	178,407	581	0.3	1
5SP	178,407	581	0.3	1
6SP	178,407	581	0.3	1
7SP	55,700	86	0.16	1
8SP	192,435	329	0.17	1
9SP	160,450	140	0.09	1
10AR	26,300	0	0	1
11AR	26,300	0	0	1
12AR	26,300	0	0	1
13AR	26,300	0	0	1
14AR	26,300	0	0	1
15AR	1,963,200	6,226	0.32	1
16AR	1,963,200	6,226	0.32	1
17AR	1,963,200	6,226	0.32	1

Table 8 Life cycle inventory values ($LCI_{Desertification}$) for the assessed plots

Plot code	WW, $LCI_{Desertification}$	RS, $LCI_{Desertification}$	*D*, $LCI_{Desertification}$	Fallow soil, $LCI_{Desertification}$
1SP	0	0	n.a.	0
2SP	0	0	n.a.	0
3SP	0	0	n.a.	0
4SP	7.6	7.6	n.a.	7.6
5SP	5.0	5.0	n.a.	7.0
6SP	7.6	7.6	n.a.	7.6
7SP	5.0	5.0	n.a.	6.0
8SP	5.0	5.0	n.a.	5.0
9SP	5.0	5.0	n.a.	6.0
10AR	n.a.	7.0	n.a.	7.0
11AR	n.a.	7.0	n.a.	7.0
12AR	n.a.	7.0	n.a.	7.0
13AR	n.a.	7.0	n.a.	7.0
14AR	n.a.	7.0	n.a.	7.0
15AR	n.a.	n.a.	5.0	5.0
16AR	n.a.	n.a.	5.0	5.0
17AR	n.a.	n.a.	5.0	5.0

n.a. not assessed

more than 4,000 weather stations over the period 1960–1996. For the Argentinean plots (10AR to 14AR), the *R* factor was estimated following the Vich (2007) method by means of Eq. (2):

$$R = 0.75 \times q^{1.197} \tag{2}$$

where *q* is the average annual recorded rainfall and the exponent is an error correction factor for the adopted regional model (Department of Agriculture and Weather Contingency 2011).

K factor It was estimated by applying Eq. (3) (Wischmeier and Smith 1978):

$$K = \frac{2.1 \times M^{1.14} \times (12 - OM) \times 10^{-4} + 3.25(s-2) + 2.5(p-3)}{100} \times 0.1317 \tag{3}$$

where *K* is the erodibility factor (t ha h ha^{-1} MJ^{-1} mm^{-1}), *M* is (percent silt) × (100 % clay), OM is the percentage of organic matter, *s* is the structure code, and *p* is the profile permeability class. Input data for applying Eq. 3 on plots 1SP to 6SP were obtained from DARP-ICC (1993) and Trueba et al. (2000). For the other Spanish and Argentinean plots, soil texture and structure data were obtained from direct field observation and soil sample laboratory analysis. For all soils containing less than 70 % silt and very fine sand, the *K* factor can be determined by applying Eq. (3) (Wischmeier and Smith 1978).

LS factor Equation (4) (Wischmeier and Smith 1978) was applied to estimate this factor:

$$LS = \left(\frac{x}{22.13}\right)^m \times \left(0.065 + 0.045s + 0.0065s^2\right) \tag{4}$$

where LS is the topographic factor (dimensionless), *x* is the slope length (in meter), *m* is a constant that depends on the plot slope gradient, and *s* is the slope of the plot (in percent). The slope length and gradient for plots 1SP, 2SP, 3SP, 7SP, 8SP, and 9SP were obtained overlapping on a GIS software, the created point layer of agricultural plots with a topographic and land use data layer (IDECyL 2009; MARM 2009c). For 4SP, 5SP, and 6SP, these data came from DARP-ICC (1993). The LS factor for the plots 10AR to 14AR came from personal communication by Engineer Silva Colomer.

C factor It was estimated applying Eq. (5). This method has been previously applied by several authors in areas with different climates: for example, by Morgan (1997), in an area with a rainforest climate, and by Van der Knijff et al., (1999) in an area with a Mediterranean climate. As shown in Eq. (5), an individual *C* value for each period (crop stage or month, C_i, ranging from 0 to 1) was weighed with the erosivity of rainfall for that period (R_i), and the annual *C* factor (C) was obtained by the addition of the partial *C* values. The overall crop stages from sowing to harvest plus the residue cover and the rough fallow periods were taken into account in the calculation of the C factor of each crop. A thorough explanation of the method applied to generate new C factors is described in Núñez et al. (2013).

$$C = \sum C_i \times R_i. \tag{5}$$

P factor A value of 1 was assigned to all plots and crops, supposing thus that any support cropland practice (e.g., contour tillage, terrace systems) is applied. In Spain and Argentina, farmers hardly apply conservation practices in soils where energy crops and cereals grow. As these crops yield low incomes and farmers' investments are as low as possible, the derived profits are still economically viable. Table 5 shows the estimated soil losses due to water erosion in each plot when it is occupied with rapeseed, wheat, digit grass, and under the reference situation without production. Table 5 also contains the related V_{Erosion} figures used in the LCI. As it can be seen, soil losses under fallow conditions are always higher than those occurring with any crop. Due to the lack of plant cover under fallow conditions, raindrops impact directly on the unprotected soil, and particles are effortlessly detached and removed.

Erosion of plots 15AR, 16AR, and 17AR were directly taken from Panebianco and Buschiazzo (2007) because we did not have the possibility to take soil samples nor to measure variables to calculate erosion rates by applying USLE like in the other plots. The mentioned authors had applied an erosion model to estimate soil erosion rates and we counted on that information and data.

The Spanish plots with the highest erosion rate were 4SP and 6SP, where high intense and energetic rainstorms (R factor) impact on steep slopes of 7.5 % (LS factor). The Argentinean plots present lower erosion rates than the Spanish ones in the common reference situation of bare soil, mainly because of the lower steepness of the terrains where the plots were located, lower than 1 %.

3.1.5 Aquifer overexploitation variable

Following the methodology proposed in part 1, a hydrological balance of the aquifers located in the area under study was carried out to derive $V_{\mathrm{Aquifer\ overexploitation}}$ values. Figures that can take this variable in the LCI are based on the water stress thresholds suggested by Alcamo et al. (2000). In the Spanish plots, different levels of groundwater stress were identified. Situations of high and medium stress occurred, as a general rule, in the groundwater masses below plots 1SP to 6SP. Aquifers under plots 1SP, 2SP, and 3SP are quantitatively and qualitatively threatened by marine intrusion, salinity, nitrate and ammonia concentration, and extraction of dry goods (ACA 2005). Sources of groundwater stress in the aquifers of the region for 4SP, 5SP, and 6SP are mainly urban development, dumping from industrial and sewage treatment plants and intensive agriculture practices (ACA 2005). Finally, groundwater bodies under plots 7SP, 8SP, and 9SP are not presently threatened by agriculture, livestock, and industrial or urban pressures (CHDuero 2005).

In Mendoza, in the artificial oasis in Mendoza where the plots (AR 10 to 14) are located, there are approximately 16,000 private drillings which complement the supply of irrigation surface water when it is not enough to satisfy the demand. Most aquifers, with a depth that varies between 100 and 200 m, are highly contaminated by salinity, rendering them unsuitable for use as agricultural irrigation water. Thus, they are rarely overexploited (Alvarez and Villalba 2003). The General Department of Irrigation, in the Master Plan of the North Basin of Mendoza, dated 2008, reports a slight deficit of 32 $\mathrm{hm}^3\ \mathrm{year}^{-1}$, but the unconfined aquifer is relatively in balance with the level of exploitation (DGI 2008). So, there is poor aquifer overexploitation and only in periods of severe scarcity of surface water. In the case of plots located in San Luis, there is no aquifer overexploitation because water comes exclusively from a surface source (personal conversation with Carlos Frasinelli).

Table 6 shows figures of the water exploitation ratio, as well as the $V_{\mathrm{Aquifer\ overexploitation}}$ values for the LCI of each studied plot.

3.1.6 Fire risk variable

The procedure to work out $V_{\mathrm{Fire\ risk}}$ required the quantification of the accumulated percentage of surface affected by forest during the latest 10 years in the geographical area under study, being that of a regional administrative division with an area size of the district (e.g., *departamento*s and *comarcas* in Argentina and Spain, respectively, Núñez et al. 2010). Results for this variable (see Table 7) show that none of the areas where the Spanish and the Argentinean plots were located have high risk of forest fires (Generalitat de Catalunya 2008; Junta de Castilla y León 2009b; Argentinean National Secretariat of Environment 2000–2010).

3.1.7 Inventory value

Following the method of part 1, the LCI value for each plot is calculated applying Eq. (6).

$$\begin{aligned} &\text{If } V_{\mathrm{Aridity}} \leq 0,\ \mathrm{LCI}_{\mathrm{Desertification}} = 0 \\ &\text{If } V_{\mathrm{Aridity}} > 0,\ \mathrm{LCI}_{\mathrm{Desertification}} = V_{\mathrm{Aridity}} + V_{\mathrm{Erosion}} \\ &\qquad + V_{\mathrm{Aquifer\ overexploitation}} \\ &\qquad + V_{\mathrm{Fire\ risk}} \end{aligned} \tag{6}$$

where $\mathrm{LCI}_{\mathrm{Desertification}}$ is the desertification value for the inventory phase and V_{Aridity}, V_{Erosion}, $V_{\mathrm{Aquifer\ overexploitation}}$, and $V_{\mathrm{Fire\ risk}}$ are the individual figures for each variable reported in Tables 2, 3, 4, 5, 6, and 7. The other elementary flows of the LCI—time and area of occupation—will be used in the LCIA phase to calculate the desertification impact of each plot.

According to Eq. (6), the $LCI_{Desertification}$ value of the assessed plots ranges from 0 to 7.6 (of a maximum of 10), see Table 8.

In Spain, $LCI_{Desertification}$ figures of plots 1SP, 2SP, and 3SP are 0. This means that these plots were located in an area with a humid climate—namely in a humid–subhumid climate—without desertification risk (United Nations 1994). Therefore, as a result of applying the model suggested in part 1, the desertification impact of the activity should not be integrated in the LCA study. The Spanish plots with higher $LCI_{Desertification}$ value were 4SP, 5SP, and 6SP because of the high soil losses by water. Comparing crops, the result of $LCI_{Desertification}$ was the same for winter wheat and rapeseed in all the plots.

In the Argentinean plots, the higher values of $LCI_{Desertification}$ were obtained for plots located in Mendoza (AR 10 to 14) because of the high aridity index. Meanwhile, desertification impact of digit grass plots is comparable to those of rapeseed and some with winter wheat from Spain.

The common reference situation of unproductive soil in fallow conditions showed two different trends when comparing plots located in Spain and Argentina: while in Spain, given a specific plot, the greatest impacts were generally for the reference situation, in Argentina, it recorded exactly the same $LCI_{Desertification}$ value than rapeseed and digit grass, although in some plots soil erosion was higher than in the cultivated plots due to the lack of vegetative cover. Results are summarized in Table 8.

3.2 Impact assessment: characterization factors and desertification impact

3.2.1 Characterization factors

Because the effect of land use depends on many site-specific conditions connected to climate and soil properties, the resulting environmental impact of a given activity should be presented relative to the biogeographical situation where it takes place, i.e., impact should be regionalized. In the desertification model presented in part 1, CFs were regionalized for the main natural areas of the world. The ecosystem classification selected was Bailey's ecoregion classification system (Bailey 2002). In part 1, the CF for a given ecoregion was calculated as the sum of the CFs for each of the four variables of the LCI. For each variable, the CF was derived from surface statistics obtained from the overlap of the ecoregion layer and the corresponding variable layer. Of the 15 Bailey's ecoregions, 7 have a characterization factor of 0 because they have, on average, a humid climate. The other eight, ecoregions have a CF that varies from 4.0—the least susceptible ecoregion to suffer desertification—to 7.6, in the most risky areas (of a maximum of 10). Further details of the method used to figure out CFs can be found in part 1.

In the case studies, Bailey's ecoregion of each plot was identified by overlapping the data layer containing the georeferenced location of the studied plots with Bailey's ecoregion layer in the GIS software. With this method, we found that the Spanish plots were located in the Mediterranean ecoregion (CF=6.3), while the Argentinean plots were in the tropical/subtropical steppe (CF=6.3) and tropical/subtropical desert ecoregion (CF=7.6), being this latter the ecoregion with the greatest desertification risk.

3.2.2 Desertification impact

Once every variable and parameter to quantify the impact of land use in dry lands is calculated, the model proposed in part 1 can be applied (Eq. (7)):

$$LCIA_{Desertification} = \left[LCI_{Desertification} \times CF^{i}\right] \times \frac{Area_{LCI\ activity}}{Log\ Area_{Ecoregion\ i}} \times t. \tag{7}$$

Apart from the $LCI_{Desertification}$ values, already reported in Table 8, figures for the remainder variables of each plot are summarized in Table 9.

The resulting desertification impact applying Eq. (7) for the 17 plots of the case studies is compiled in Table 10.

Of all the plots, the greatest desertification impact was in plots 4SP and 6SP, where the same impact result for the three systems studied was recorded: both crops and the unproductive fallow soil ($LCIA_{Desertification}=0.074 \times km^{2}_{plot}\ km^{-2}_{Ecoregion} \times year$). The result means that rapeseed, wheat, and bare soil can potentially lead to the desertification of the same area of the ecoregion, being this of 0.074 km^2 (74 ha) in 1 year. In plots 1SP, 2SP, and 3SP, which recorded a value of 0 in the LCI, not being an arid climate, the result of LCIA was also 0, thus having no impact of desertification. When comparing the impact produced by the crops against the baseline bare soil, only 3 of the 13 plots showed a desertification impact worse than fallow plots (5SP and 9SP 7SP), while the same impact was obtained for the remaining plots.

4 Discussion

On the basis of the obtained results, it can be inferred that cultivating 1 ha of rapeseed and 1 ha of wheat has the same impact on the analyzed plots in Spain and improves the reference state conditions in 50 % of cases. Moreover, rapeseed grown in the analyzed plots in Mendoza (10AR–14AR) produces almost the same impact as when grown in

Table 9 Input parameters required to calculate $LCIA_{Desertification}$

Plot code	CFi[a] (−)	Area LCI activity (Km^2)	Log area ecoregion[b] (Km^2)	Time activity (years)
1SP	6.3	0.01	6.45	1
2SP	6.3	0.01	6.45	1
3SP	6.3	0.01	6.45	1
4SP	6.3	0.01	6.45	1
5SP	6.3	0.01	6.45	1
6SP	6.3	0.01	6.45	1
7SP	6.3	0.01	6.45	1
8SP	6.3	0.01	6.45	1
9SP	6.3	0.01	6.45	1
10AR	7.6	0.01	7.25	1
11AR	7.6	0.01	7.25	1
12AR	7.6	0.01	7.25	1
13AR	7.6	0.01	7.25	1
14AR	7.6	0.01	7.25	1
15AR	6.3	0.01	7.14	1
16AR	6.3	0.01	7.14	1
17AR	6.3	0.01	7.14	1

[a] See part 1 for further details

some of the Spanish plots of the Noguera district (4SP, 6SP), and in both cases, the impact is greater than in the other studied areas of Spain (1SP–3SP and 7SP–9SP).

Three of the plots located in Spain (1SP, 2SP, and 3SP) resulted in having no desertification impact. There, according to local statistics, the aridity index corresponded to a humid–subhumid climate ($V_{Aridity}=0$), without desertification risk (see Table 2). At the same time, these plots were within a dry ecoregion, the Mediterranean, with an average dry–subhumid climate and therefore threatened by desertification if land use is unsustainable. This means that not always the use of a piece of land in an ecoregion with a desertification hazard will lead to negative effects on the dry land. Effects will depend on the specific location of the activity. On the contrary, if the climate had been arid, semiarid, or dry–subhumid ($V_{Aridity}>0$) but within an ecoregion with no desertification risk (CF=0), the proposed model (Eq. (7)) would have identified the activity as not leading to desertification damage. In

Table 10 $LCIA_{Desertification}$ ($km^2_{plot}\times km^{-2}_{Ecoregion}\times year$) for the evaluated plots

Plot code	$LCIA_{Desertification}$, WW	$LCIA_{Desertification}$, RS	$LCIA_{Desertification}$, D	$LCIA_{Desertification}$, fallow
1SP	0.000	0.000	n.a	0.000
2SP	0.000	0.000	n.a	0.000
3SP	0.000	0.000	n.a	0.000
4SP	0.074	0.074	n.a	0.074
5SP	0.049	0.049	n.a	0.068
6SP	0.074	0.074	n.a	0.074
7SP	0.049	0.049	n.a	0.059
8SP	0.049	0.049	n.a	0.049
9SP	0.049	0.049	n.a	0.059
10AR	n.a	0.073	n.a	0.073
11AR	n.a	0.073	n.a	0.073
12AR	n.a	0.073	n.a	0.073
13AR	n.a	0.073	n.a	0.073
14AR	n.a	0.073	n.a	0.073
15AR	n.a	n.a	0.044	0.044
16AR	n.a	n.a	0.044	0.044
17AR	n.a	n.a	0.044	0.044

n.a. not assessed

both cases, $LCIA_{Desertification}$ is equal to zero and desertification impact should not be integrated in LCA studies.

If instead of calculating the desertification impact of a standard area such as 1 ha, the real area of each plot is considered, the most affected by desertification impact would result in 5SP and 9SP in Spain and 15AR and 17AR in Argentina. This situation can be explained because the occupied area in these plots is bigger than in the other plots (90 and 138 ha, respectively) and the area of the plot is a direct influencing variable in the impact model over the desertification impact associated with an activity. As a consequence of the latter, normalized areas of plots could be useful to compare results in different regions. However, it is not advisable to normalize the occupied area when the environmental profile of a specific product, process, or activity is being assessed by a life cycle assessment study because the result will not represent reality.

The indicator system integrates different aspects of the desertification mechanism in a semiquantitative approach. The range of classification used for each variable was set up taking into account the relative importance that each variable has in triggering desertification (e.g., soil erosion values range from 1 to 3, whereas aquifer overexploitation from 1 to 2, which means that soil erosion has major impact). However, there are no scientific base proofs which explain the connection between the ranges of classification used and the actual severity, e.g., for soil erosion >25 t/ha year, $V_{Erosion}=3$ and thereby two times larger than erosion <12 t/ha year, $V_{Erosion}=1$. The suggested semiquantitative classification has definitely an impact on the results obtained. Because of this, we tested the ranges of classification making them more sensitive to the actual severity, which is creating more intensity categories. For example, considering four erosion categories such as <7/7–12/12–25/>25, inventory results do not vary with significance differences. In consequence, CFs do not vary either. Modifying the other variables, similar results were found. The other variables are not high susceptible of changes because the definition itself is in that way for each case, for example, aridity index is defined in the same ranges as UNCCD (1994) defines it.

Even if it is not shown in this paper, other calculations with different crops were done in the Mendoza Region, like garlic (*Allium sativum*), vine (*Vitis vinifera*), and Jerusalem artichoke, also called topinambour (*Helianthus tuberosus*), a summer energy crop, that could be irrigated with sewage, enabling a higher production and also avoiding competition for the use of water (Rébora 2008). In the three cases, the same result as when considering rapeseed was obtained. This could be explained because as was told in previous sections, in very arid regions, like Mendoza, the aridity index is the variable with the most weight showing its high influence and pressure in dry lands and desertification impact. As a conclusion, the aridity index is the variable in the desertification impact model that can never be avoided. On the other hand, the most sensitive variables in the desertification phenomenon are region aridity and soil erosion. This is important because, in the extreme situation of fire risk or aquifer overexploitation data not being available, they could be set aside of the model and only consider aridity and soil erosion as having a good approximation of desertification impact of a product, process, or activity. Finally, the model has the ability to include other relevant variables according to the case and location desertification vulnerability. The only condition for this is that when performing a comparative study all the alternatives should be analyzed using the same variables, both in the inventory and in the impact assessment stages.

The model proposed needs for specific data for assessing a specific product, process or activity that in some opportunities could seem difficult to get. However, in the most cases information related to the considered variables is available in general datasets like meteorological data, soil parameters, fire risk information, water balances among other, and only few times measures in field campaigns need to be performed.

Working with spatial data in the LCI and the LCIA involves spatial uncertainties that might have a strong impact on the indicator result. In the LCI, spatial uncertainty is linked to inaccuracies in both the location of the individual data point and the underlying data sources used to identify the environmental conditions at the given location. Usually, the quality level of the inventory data depends on the spatial resolution of the georeferenced database used: the greater the spatial detail it has, the lower the spatial uncertainty of the LCI. Therefore, it is a priori more convenient using local or regional information sources than information on larger spatial scales. CFs from the LCIA method have been derived gathering information of the same variables of the LCI from maps and data sources resolved at the global level and regionalizing the information at the ecoregion level (see part 1). These maps have also inherent spatial uncertainty that add statistical uncertainty to results. Finally, the regionalization of continents into ecoregions instead of, e.g., countries, determines the quantity and spatial differentiation of CFs.

The feasibility of the model is demonstrated by quantifying the impact of desertification in the case study. Moreover, by applying it in agricultural plots located in different continents, the model has allowed understanding the degree of difficulty in obtaining the data needed to be collected in the inventory stage. In both Spain and Argentina, two countries with different systems of information management, data were obtained from data sources at state level. A lot of the specific data of each soil type and those relating to crop management has been surveyed on site. The latter may mean in some cases some difficulty of access to information, as this is conditioned by the willingness and availability of the producer. In order to assess the data quality, we recommend consulting the guidelines suggested by the International Reference Handbook Life Cycle Data System (European Commission 2010), points 1 to 6.

There was an even greater difficulty in obtaining information related to the exploitation of aquifers in Mendoza and San Luis. The model proposed is an LCI data intensive model, and it is therefore difficult to apply in developing countries, where there are usually very few reports on environmental conditions and there is some difficulty of access to information.

Regarding all this, the authors propose further research to relate the affected area desertified to the potentially affected species depending on the specific ecoregion, aiming at obtaining an endpoint.

5 Conclusions and recommendations

The results obtained in the case studies demonstrate and confirm the working hypothesis at the beginning of the investigation "in drylands it is necessary to have a land use model to specifically address the characteristics of this type of land and be applicable overall arid regions of the planet." The proposed model implies a contribution of significant importance because so far there has not been an impact assessment tool for land use in dry lands within the LCA framework. With this work, it was found that the main strength of the proposed model is that it allows a simple way to quantify the desertification impact. On the other hand, we emphasize that the model can be adapted virtually without difficulty to the evaluation of all types of crops with different management practices in different regions with a really simple inclusion in the life cycle impact assessment stage. It is also highlighted that the desertification model proposed may present some operational difficulties when applied in case studies other than agricultural. For example, when a building material like clay bricks is assessed in a dry land (in an LCA with a functional unit defined as 1 m^2 of clay brick wall), inputs such as fire risk or aquifer overexploitation may be meaningless because in this specific case and in other similar cases, fires are not a consequence of brick-manufacturing activities, while deforestation is. Work is underway to adapt the most appropriate variables allowing the reflection on the desertification impact when applied to study the non-agricultural sector. The use of USLE should be complemented by other impact tools like sealing soil in cases of urban settlements, facilities, or other installations.

Finally, we are planning a future research which is needed to incorporate into the proposed model the transformation aspects in land use. So far, the model only deals with the impact of land occupation.

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