Land Use in LCA (Subject Editor: Llorenç Milà i Canals)

Key Elements in a Framework for Land Use Impact Assessment Within LCA

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Abstract

Background, Aims and Scope. Land use by agriculture, forestry, mining, house-building or industry leads to substantial impacts, particularly on biodiversity and on soil quality as a supplier of life support functions. Unfortunately there is no widely accepted assessment method so far for land use impacts. This paper presents an attempt, within the UNEP-SETAC Life Cycle Initiative, to provide a framework for the Life Cycle Impact Assessment (LCIA) of land use.

Main Features. This framework builds from previous documents, particularly the SETAC book on LCIA (Lindeijer et al. 2002), developing essential issues such as the reference for occupation impacts; the impact pathways to be included in the analysis; the units of measure in the impact mechanism (land use interventions to impacts); the ways to deal with impacts in the future; and bio-geographical differentiation.

Results. The paper describes the selected impact pathways, linking the land use elementary flows (occupation; transformation) and parameters (intensity) registered in the inventory (LCI) to the midpoint impact indicators and to the relevant damage categories (natural environment and natural resources). An impact occurs when the land properties are modified (transformation) and also when the current man-made properties are maintained (occupation).

Discussion. The size of impact is the difference between the effect on land quality from the studied case of land use and a suitable reference land use on the same area (dynamic reference situation). The impact depends not only on the type of land use (including coverage and intensity) but is also heavily influenced by the bio-geographical conditions of the area. The time lag between the land use intervention and the impact may be large; thus land use impacts should be calculated over a reasonable time period after the actual land use finishes, at least until a new steady state in land quality is reached.

Conclusion. Guidance is provided on the definition of the dynamic reference situation and on methods and time frame to assess the impacts occurring after the actual land use. Including the occupation impacts acknowledges that humans are not the sole users of land.

Recommendations and Perspectives. The main damages affected by land use that should be considered by any method to assess land use impacts in LCIA are: biodiversity (existence value); biotic production potential (including soil fertility and use value of biodiversity); ecological soil quality (including life support functions of soil other than biotic production potential). Biogeographical differentiation is required for land use impacts, because the same intervention may have different consequences depending on the sensitivity and inherent land quality of the environment where it occurs. For the moment, an indication of how such task could be done and likely bio-geographical parameters to be considered are suggested. The recommendation of indicators for the suggested impact categories is a matter of future research.

Keywords: Biodiversity; bio-geographical differentiation; dynamic reference situation; land quality; land use; land use impacts; LCA; LCIA; natural environment; natural resources; site-dependency; soil quality

Introduction

Land use by agriculture, forestry, mining, house-building or industry leads to substantial impacts, particularly on biodiversity and on soil quality as a supplier of life support functions. Unfortunately there is no widely accepted assessment method so far for land use impacts. Within the UNEP-SETAC Life Cycle Initiative[1], key elements in a Life Cycle Impact Assessment (LCIA) framework of land use have now been treated and are presented in this paper.

The goals of this paper are to start a dialogue with experts outside the Life Cycle Assessment (LCA) field and to provide guidelines to LCIA method developers on the key elements to be addressed when assessing impacts from land use. The discussion presented here is valid both for midpoint and damage approaches (Jolliet et al. 2004), and specific comments

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on these are made through the paper. Recommendations on specific methods and indicators are left for future publications. The framework presented here is also relevant for LCA practitioners interested in characterising impacts from land use, as it may guide them on the main issues to be covered by any particular method they use.

The need for the assessment of land use impacts is extensively justified in the literature (FAO 1976, Barrow 1991, ISRIC and UNEP 1991, EEA 1995, Pimentel et al. 1995, Müller-Wenk 1998, EEA and UNEP 2000, Chapin et al. 2000, Sala et al. 2000, COM 2002) and specific international conventions and agreements (e.g. Convention on Biological Diversity[2], UN Convention to Combat Desertification[3], Convention on the Conservation of Migratory Species of Wild Animals[4], Ramsar[5], etc.). LCA was developed as a space- and time-independent environmental assessment methodology for (industrial) product development, while other impact assessment methods (mainly Environmental Impact Assessment, EIA) were developed to assess the impacts of localising a project in a specific site. However, when LCA moves into land-use-related issues the clear borders between LCA and EIA become somewhat less distinct, and LCA needs to partially adapt the methodology from EIA and other tools. For some types of decisions EIA and other tools may be more adequate than LCA. There has been extensive debate on land use impact assessment in LCA. Two SETAC working groups (Udo de Haes et al. 1996, Lindeijer et al. 2002) started to frame the issue of land use impacts. A special issue in the Journal of Cleaner Production (issue 8: 2000, see Lindeijer 2000a), provides a review of some of the existing methods and framework for land use impact assessment. Heijungs et al. 1992 and Fava et al. 1993 basically mention land use as a source of environmental impacts, and consider it from an inventory point of view (although they measure it in m^2, failing to recognise the flow nature of land use). Many references focus on suggesting indicators to include the effects of land use on biodiversity and biomass production, although the practical implementation of such sets of indicators is seldom checked with a consistent framework (Audsley et al. 1997, Blonk et al. 1997, Cowell 1998, Mattsson et al. 1998, Müller-Wenk 1998, Lindeijer et al. 1998, Baitz et al. 1999, Köllner 2000, Lindeijer 2000b, Schenck 2001, Weidema and Lindeijer 2001, Brentrup et al. 2002, Milà i Canals 2003, Bauer and Zapp 2004, Kyläkorpi et al. 2005, Jeanneret et al. 2006, Oberholzer et al. 2006). Some of the latest methods for LCIA thoroughly address land use impacts, but fail to consistently address all the main impact pathways and/or include the effects from occupation and transformation interventions; Goedkoop and Spriensma et al. (1999) offer a sophisticated method to assess the effects from land use on biodiversity ('ecosystem quality') on a damage level, but fail to address the effects on the resource aspect of land; Guinée et al. (2002) suggest using the LCI results of m^2year as a baseline approach, and offer a review of methods to address impacts on biodiversity and life support functions in a more comprehensive way. The major environmental importance of land use impacts contrasts with the lack of consensus on this area within the field of LCA (Jolliet et al. 2004). As a result, land use impacts are seldom included in LCA, and the credibility of LCA results is insufficient to many stakeholders, who as a result have to address land use impacts using other tools. Their inclusion in LCIA is crucial since the production of raw materials (fibres, food, energy carriers, metals etc.) often takes place in ecologically fragile areas.

The lack of consensus comes at least partly from a lack of understanding on the goal-dependency of LCA, as well as from the failure to recognise the value judgments behind the methodological decisions for land use impact assessment. The paper addresses these value judgements in the following sections:

- What are the functions of land that need protection? (see section 1)
- Perception of 'ownership': is land for human use or do we have to share it with other users? (occupation and transformation impacts; see the environmental mechanism of land use impacts in section 2)
- Which indicators represent the necessary impact pathways? Section 3 briefly comments potential indicators for the different impact pathways.
- Assumptions on future or alternative land uses: Is the land use ever going to change? Is natural relaxation going to happen after any land use? What would be there if the studied system was not established? (see section 4.1)
- Time perspective of the assessment: do we consider infinite time for recovery? If this is the case, what is the relevance of including reversible impacts in the assessment? (see section 4.2)
- Perception of land's recovery capacity: is land robust or fragile? See e.g. discussion on thresholds in section 4.3.

Section 5 summarises the conclusions and recommendations from the paper.

1 Main Features: Description of Impacts from Land Use

Land provides support functions for life, including cycling of nutrients, water and carbon, and the provision of habitat for both human and non-human life (Teller et al. 1995, Lindeijer et al. 2002, Candinas et al. 2002, Milà i Canals 2003, p. 182). Some of land's functions have an economic consequence (habitat for humans; aesthetic and cultural value; agronomic value), and are therefore partly internalised within the economic system; these functions should be included in the LCA through the functional unit or a secondary function. On the other hand, the ecological functions of land are most often externalised from economic assessments, and should be covered in land use impact assessment. For the purposes of this paper, we refer only to the impacts on land quality[6] itself that are not covered by traditional LCIA impact categories[7]:

[2] <http://www.biodiv.org/default.shtml>
[3] <http://www.unccd.int/>
[4] <http://www.cms.int/>
[5] <http://www.ramsar.org/key_conv_e.htm>

[6] Land quality is used here in the sense of fulfilment of the land functions related to the safeguard subjects to be protected by humans. The units of measure depend on the particular functions of interest for any one user.

[7] Land use also affects many other impact categories, such as eutrophication and acidification (through e.g. removal of standing biomass); toxicity (through application of pesticides); biotic resource depletion (e.g. extraction of wood from natural forests); etc. These impacts should be addressed in the relevant impact categories, and will not be the subject of discussion in the present paper.

- impacts on the existence (or intrinsic) value of biodiversity[8], seen as a key element of the biotic natural environment,
- impacts on biotic production potential (including soil fertility and use value of wild species, e.g. for agriculture), which is a key element of the semi-biotic[9] natural resources exploited by humans,
- impacts on ecological soil quality (including other life support functions of soil: filter and buffer capacity, water carbon and nutrients cycling).

1.1 Effects of land use on the existence value of biodiversity: impact on the natural environment

Land use is related to important changes in species composition on and around the used area, e.g. when a prairie is ploughed to provide space for arable agriculture (a case of land transformation) the species composition is severely changed and reduced. Land occupation maintains a species composition different than the one that would be there without the studied land use (Müller-Wenk 1998, Lindeijer et al. 2002). Other indirect effects on biodiversity also occur through alterations in the soil functions (Weidema 2002), which may affect the species composition.

1.2 Effects of land use on biotic production potential: impacts on the natural resources

Soil quality may be generally defined from the performance of life support functions (Milà i Canals 2003), which roughly include biotic production; substance cycling and buffer capacity; climate regulation (Udo de Haes and Lindeijer, 2002, pp. 220–221). Biotic production is the main soil function directly used by humans, and can be therefore defined as a natural resource aspect of soil. Fertile land is used every year again as an input into man-controlled food, fuel and fibre production processes. Unfortunately, this 'flow-type' resource may be deteriorated by certain uses. E.g. the use of heavy machinery in agriculture or forestry may lead to soil compaction and reduction of soil porosity, thus disturbing rainwater infiltration; water holding capacity; root development; etc. (physical soil fertility). Leaving a bare soil in critical times of the year may also lead to increased topsoil erosion, and thus to the loss of the most fertile part of the soil (loss of chemical fertility). An extreme case of fertility depletion would be building on fertile land; in this case the resource is completely lost (during the occupation).

In addition, the production of useful biomass requires appropriate species (natural or bred/modified by humans). If the effects of land use on species which are useful for humans (i.e. with a use value) can be modelled, it is advisable to include this effect in the impact pathway 'biotic production potential'. This may be particularly relevant in agricultural systems (Swift and Anderson 1994), e.g. for pollination and biological pest control.

1.3 Effects of land use on ecological soil quality: impacts on the natural environment

Soil quality is not only affected when fertile land is used, and biomass production is not the only life support function to be protected. Soil quality as an element of the global water, carbon[10] and nutrients cycles, and as a filter and buffer of hazardous chemicals also needs to be protected. This is relevant for any type of land being used, and not only for fertile land as a resource. E.g. when a meadow is sealed to build a road, and during the use of land as a road, the soil is sealed, affecting its role in the water cycle and possibly generating off-site impacts such as increased surface runoff and flooding in neighbour areas.

2 Results: Environmental mechanism – Occupation and Transformation as Processes, Interventions and Impacts

From a Life Cycle Inventory (LCI) perspective the term *occupation process* refers to the use of a land area for a certain human-controlled purpose (agriculture, waste dumping, building, etc.), assuming no intended transformation of the land properties during this use (Lindeijer et al. 2002, p. 40 and Fig. 2–2). In contrast, a transformation process implies the change of a land area according to the requirements of a given new type of occupation process (e.g. draining a marshy area for its subsequent use as cropland) (Lindeijer et al. 2002, p. 41 and Fig. 2–3). If land use processes are listed in LCI, it is usual to call them interventions or elementary flows, similar to all other types of LCI entries. *Occupation interventions* are measured in surface-time units (e.g. ha yr), representing a certain area of land of a given type (e.g. 1 ha of grassland) used over a certain time period (e.g. 2 yr). On the other hand, *transformation interventions* are measured in surface units (e.g. 2 ha of grassland converted into road).

If a transformation process of natural land is not followed by any land occupation processes so that the land lies fallow after transformation, the sudden change of land quality due to the transformation process will be followed by a gradual reversal of the initial quality change, due to the forces of nature (**Fig. 1**). The land quality will in general become roughly equivalent to the pre-transformation state after years or decades or centuries, depending on the severity of transformation, the site's bio-geographical conditions and the size of the area considered. The *transformation impact* is represented by the shaded area between the dotted line (reference

[8] It may be noted that effects on biodiversity (through damages on biotic environment, or occurrence of species) are currently considered in LCA (Jolliet et al. 2004); however, the effects considered have been traditionally limited to those caused by changes in the chemical composition of the environment (toxicity; eutrophication; acidification).

[9] It should be noted that it is somehow arbitrary to classify soil as 'biotic' or 'abiotic'; it is actually an interface between the biotic and abiotic environment, and contains both biotic and abiotic resources. Even though soil has often been classified under 'abiotic resources' it is here suggested to call it 'semi-biotic'.

[10] Land transformations may have a big effect on the role of land as a source or sink of CO_2 emissions, with natural land generally acting as a sink of carbon: IPCC (2001) estimates that greenhouse gas emissions caused by land cover changes between 1850 and 1990 are of the same order of magnitude as those derived from combustion (121 Pg C in front of 212 Pg C). However it has not been common practice in LCA to consider the role of soil organic matter in the greenhouse effect (with some exceptions: Kim and Dale 2005, Svensson 2005; Coltro et al. 2003 consider carbon emissions from flooded areas).

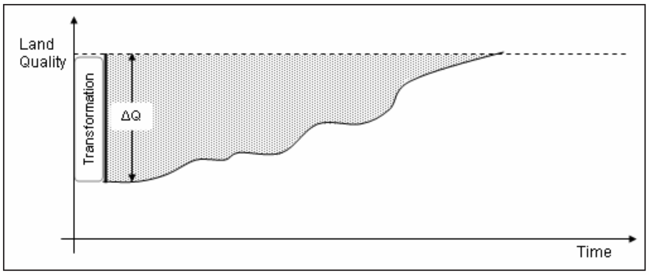

Fig. 1: Pure case of land use by a transformation process not followed by any occupation process. ΔQ represents the initial change in land quality, and the shaded area represents the transformation impact

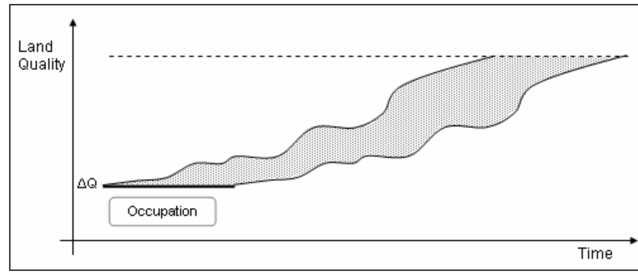

Fig. 2: Pure case of land use by an occupation process with no land quality change during occupation (ΔQ=0). The original transformation process was somewhere in the past. The shaded area represents the occupation impact

situation) and the full line (studied system) in Fig. 1, i.e. the integral of ΔQ over time. The magnitude of this impact may be coarsely approximated by a triangle. The figure assumes that land quality would not have changed without the studied system, and so the reference situation (see section 4.1) is the initial state of land quality.

It must be noted in Fig. 1 that land quality may actually be represented by different parameters, depending on the specific impact pathway (e.g. biodiversity; biotic production potential; ecological soil quality), and that the same land use might actually be damaging for biodiversity and beneficial for soil quality, and vice versa.

If a land occupation process does not cause any sizeable quality change of the occupied land, it may nevertheless cause an impact, because the forces of nature are prevented from changing the land qualities during the occupation time. In other words, the spontaneous rebound of land quality is postponed by a period of time equal to the duration of the occupation process (**Fig. 2**). The *occupation impact* is represented by the shaded area between the two full lines, which is the difference of the two integrals of ΔQ over time. The magnitude of this impact may be coarsely approximated by a rectangle ΔQ times the duration of the occupation process, assuming that relaxation before and after the occupation process follow parallel lines. In this case, the land relaxation without studied system is used as a reference situation (see section 4.1).

In reality, land use generally consists of a mix of the two archetypal pure cases: A transformation process is normally followed by one or many occupation processes, and an occupation process is accompanied by comparatively small changes in quality. This is potentially relevant when different intensities of a process in the context of the same occupation are compared (for example an extensive versus an intensive used permanent meadow, leading to differences in e.g. fertilising level, use of machinery, pesticide dosage, etc.). Transformation and occupation impacts may be added because both represent a land quality difference during a certain time. Considering a mixed situation between Fig. 1 and 2, **Table 1** presents a possible series of human interventions and events and their potential effects on land quality. The solid line in **Fig. 3** represents this possible evolution of land quality (y-axis) in time (x-axis), as affected by the human interventions (transformation and occupation processes) at each time stage. In contrast to Fig. 1 and 2, all quality changes in Fig. 3 are shown as linear, for the sake of simplicity.

The changing land quality between t_0 and t_{rel} in Fig. 3 might actually be represented as a 'staircase' of pure occupation and transformation periods, where land quality was constant during the occupation and changed abruptly in the

Table 1: Likely effects of human interventions during a land use

Time stage	Human interventions	What happens to land quality (Q)
Before t_0	No human intervention: A certain land cover is in place (e.g. grassland) in a steady state	Slow natural evolution; this is depicted as a static state for simplicity, but natural fluctuations in Q are likely to occur
t_0 (transformation)	Transformation process: Humans transform land to make it suitable for a new use (e.g. grassland is ploughed to be used as cropland)	Land quality changes briskly from Q_{his} to Q_0. A decrease is shown in the figure, but an increase in land quality may also take place due to human interventions. Besides, land quality may increase for some of the impact pathways considered while decreasing for others
t_0 to t_{fin} (occupation)	Occupation process: The land area is used for the new use (e.g. cropping)	Land quality gradually evolves under the new land use, from Q_0 to Q_{fin}. Again, a decrease is shown here for simplicity, but more complex evolution is likely, depending on the land management practices
t_{fin} to t_{rel} (relaxation time)	No human intervention: Spontaneous change of land quality due to forces of nature (natural relaxation)	Land quality changes from Q_{fin} to Q_{rel} (e.g. biodiversity and soil quality increase as natural succession takes place after cropping land for some years; this process could also be accelerated by human induced restoration, e.g. afforestation). The figure shows a relaxation period shorter than the occupation (e.g. 20 years of recovery after 150 years of agriculture), but other situations may be expected (e.g. 2 years of recovery after a very intensive year of cropping)
After t_{rel}	A new land use is in place (e.g. unused land)	If land is left undisturbed land quality probably reaches a new steady state after some decades to centuries (relaxation time); otherwise, quality evolution depends on the new land use

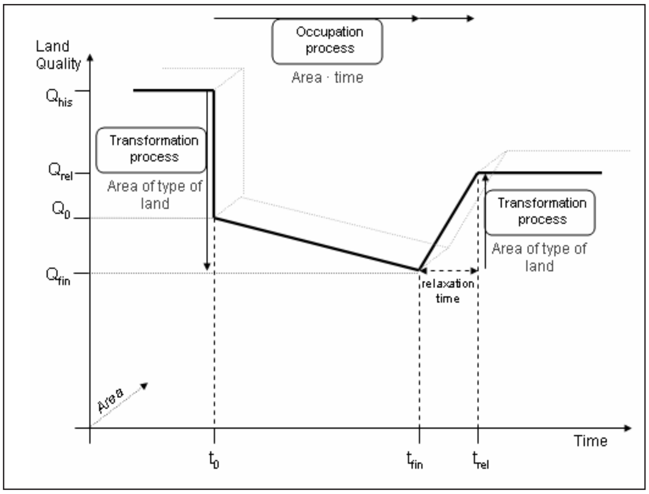

Fig. 3: Evolution of land quality with land use interventions (adapted from Lindeijer et al. 2002)

transformation. Integrating these steps over time (i.e. making the occupation periods more and more small) a continuous line like the one depicted in the figure is obtained. The z-axis represents the area affected, as the land use impacts are proportional to the area of used land, but this will not be further shown in future figures for clarity.

In summary, the LCI needs to record the three dimensions affected by land use:

- Area (surface used)
- Time (duration of the occupation and transformation processes)
- Quantitative description of the occupation and transformation processes in order to quantify land quality and the reference situation before, during and after the land use

3 Indicators for Land Quality

In the LCIA, impacts on the relevant impact categories (at the midpoint or damage level) from the occupation and transformation aspects of land use need to be assessed using suitable indicators. It needs to be stressed again that land quality should be measured in different units for the different impact pathways affected. The impact indicators for each of these impact pathways require information on different parameters from the LCI in order to be calculated. Extensive work still needs to be done for the definition of such impact indicators, which is the focus of ongoing work within TF2. **Table 2** is a non-exhaustive list of potentially useful indicators for each of the land use impact pathways, with likely requirements in terms of LCI parameters.

There are many other references for indicators at midpoint level. For impacts on biodiversity see also Cowell (1998), Köllner (2000). For biotic production potential, see e.g. Feitz and Lundie (2002), Milà i Canals et al. (2006). Sets of midpoint indicators for the biotic production potential and ecological aspects of soil quality are suggested e.g. by Cowell

Table 2: Examples of possible indicators at midpoint and damage levels for the described impact pathways from land use, including requirements of LCI information

Impact pathway	Indicator	Level	LCI modelling aspects
Biodiversity (intrinsic value) – Natural environment	PDF [a] or PAF [b]	Damage	It may be fruitful to work with these indicators as they are currently used by eco-toxicity categories (Jolliet et al. 2004). However these indicators do not reflect other important aspects like the relative scarcity of species
	% of threatened vascular plant species in region	Midpoint	Description of the land use interventions to render possible a link to empirical data on number of vascular plant species per km^2 (Müller-Wenk 1998)
	Red-listed species; key features	Midpoint	Species correlation with habitat, the ecological habitats found and affected area (Kylakörpi et al. 2005)
	Global species diversity; nature protection	Midpoint	Effects of agricultural activities (e.g. nitrogen flows; number of grass cuts; etc) on eleven groups of indicator species (Jeanneret et al. 2006)
Biotic production potential – Natural resources	Surplus energy (+ possibly other interventions) [c]	Damage	Requirements to restore soil quality through e.g. addition of organic amendments and other soil fractions (clay; sand); other interventions may include e.g. gaseous emissions from organic amendments (Milà i Canals et al. 2006)
	Deficit of Soil Organic Matter (SOM) [Mg SOM year]	Midpoint	Changes in SOM due to the studied system, which may be obtained by different means (Milà i Canals et al. 2006); additions of organic matter (e.g. manure; crop residues); effects of agricultural practices on degradation rates
	Eroded soil [kg soil lost]	Midpoint	Measured or calculated with empirical or contextual models of the soil-erosion process (e.g. USLE[d] or SLEMSA[e]), requiring slope gradient; rainfall intensity; vegetation cover; soil type
Ecological soil quality – Natural environment	To be explored, according to the affected impact pathways	Damage	To be explored, according to the affected impact pathways (e.g. global warming, toxicity…)
	Combinations of 9 indicators: pore volume; SOM (see above); microbial activity; etc.	Midpoint	Effects of agricultural activities (e.g. heavy metals flows; preceding and following crop; etc) on nine soil quality indicators (Oberholzer et al. 2006)

[a] Potentially Disappeared Fraction of species
[b] Potentially Affected Fraction of species
[c] the impacts from land use-based activities are not properly represented by energy indicators (Walk et al. 2005, Huijbregts et al. 2006) and therefore the 'surplus energy' indicator should be combined with other emissions related to the restoration activities, leading to other damages
[d] Universal Soil Loss Equation (Wischmeier and Smith 1978)
[e] Soil Loss Estimation Model for Southern Africa (Elwell and Stocking 1982, Elwell 1984)

(1998), Mattsson et al. (1998), Baitz et al. (1999). Milà i Canals (2003) suggests a simplified approach with soil organic matter as a single indicator for life support functions. Biomass production (Lindeijer et al. 1998, Weidema and Lindeijer 2001) or yield gap indicators (Bindraban et al. 2000) are inadequate indicators for soil quality insofar as they express high short-term yields due to skilled agricultural management and addition of fertilisers (Burger and Kelting 1999, Bouma 2002).

It has been common practice in agricultural LCA to include 'land use' as m^2year, an inventory indicator expressing the land occupation. This is based on the assumption that 'less use of land is better', which does not consider impact pathways on the existence value of biodiversity, biotic production potential or ecological soil quality and does not allow to differentiate the impacts of a same occupation with different intensities, i.e.: less intensive uses of land may be less damaging on biodiversity and soil quality than more intensive ones.

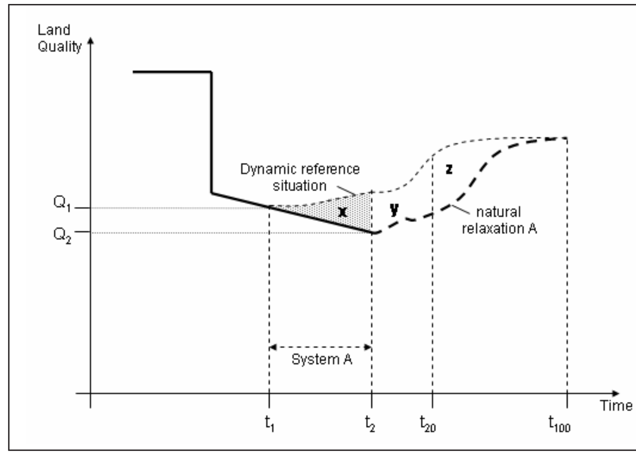

Fig. 4: Impacts during the actual land use of system 'A' using natural relaxation as the dynamic reference situation (shown by the shaded area marked with 'x'). The impacts after the end of the actual occupation process (t_2) allocated to system A depend on the time frame of the impact assessment (t_{20} and t_{100} are here depicted as example time frames); x+y= total land use impacts in a time frame of 20 yr; x+y+z= total land use impacts in a time frame of 100 yr

4 Discussion: Application of the Framework to LCA

4.1 Reference situation for land use impacts

As shown in Fig. 1 and 2, a reference is needed against which one can measure the additional damaging effects on nature caused by the studied land use (e.g. 'use of an area of size 1 km^2 for soybean production during 1 year'). The 'reference situation' without the studied land use is not obvious. Using either the historic natural land *state* (Blonk et al. 1997, Brentrup et al. 2002, Q_{his} in Fig. 3) or the potential *state* after relaxation (Lindeijer et al. 2002, Q_{rel} in Fig. 3) as a reference *state* provides information on the relevance of the impacts caused by the studied system, but does not take into account the dynamic nature of land evolution, and raises problems of allocation between successive land uses.

It is here suggested to use the term *dynamic reference situation* (baseline, in the Kyoto protocol terms), which in this case refers to the *non*-use of the area. But this raises further questions:

1. Does 'non-use' mean that the corresponding area is assumed to be totally free from any human land use during the 1 year, or is it understood that the area would be occupied for some other economic purpose?
2. If the area is not used for the studied human activity, what is then the quality evolution of this area?

The answer to the first question is related to the purpose of the LCA. If the LCA is aiming at describing the system's impacts (called retrospective, descriptive or attributional LCA), the LCA practitioner should focus on determining all the impacts caused by the studied activity (Tillman 2000), relative to a situation where this activity is not undertaken. Consequently, the adequate reference situation for retrospective LCA studies is natural relaxation (**Fig. 4**). Fig. 4 shows only one possible example; other situations might happen where the studied system increases land's quality (e.g. improvement in biodiversity as a result of good land management by the land using activity), even above the reference situation. In this case, the studied system would be credited for the improvement of land quality. The shaded area in Fig. 4 (marked with an 'x') represents the impacts during the occupation process; section 4.2 deals with the impacts occurring after the occupation process ('y+z' in Fig. 4).

On the other hand, if the study aims at evaluating the consequences of changes in land use (prospective or consequential LCA[11]), only the changes in land use impacts directly due to the studied system respect an alternative system are considered. Therefore, the alternative system becomes the reference[12].

The dynamic reference situation must be defined in the goal and scope definition, consistently with the goals of the study. If an alternative land use is used as a reference, the practitioner must provide enough evidence to proof the likeliness of such an alternative system (e.g. with statistical time series of land uses).

As for the second question, on the quality evolution of this land area when the studied activity is not established there, some expert judgement and modelling expertise will be required. Land quality evolution under natural relaxation conditions can be worked out in a two steps procedure: As a first step, the expected natural land cover or soil quality for the location of the land area is determined on the basis of global land cover or soil maps, whereby the relaxation quality (Q_{rel} in Fig. 3) of the location can be 'interpolated' from the neighbouring grid cells being in natural or near-to-natural

[11] Consequential LCA focuses on the effects of substitutions among alternative product systems (Weidema 2001), providing information on the environmental consequences of individual actions (Ekvall et al. 2005).

[12] The alternative situation may be derived from statistical time series (Ekvall and Weidema 2004) for land use and must be defined in the goal and scope of the study. Natural relaxation may also be the reference situation in consequential LCA, whenever it is the most likely alternative situation to the studied system (e.g. in countries where agricultural land is being set-aside, Milà i Canals et al. 2006).

state. The second step consists in determining the time required for the relaxation, based on expert knowledge. The relaxation time depends on the last type of occupation, the type of expected natural land cover, and on the bio-geographical conditions of the location[13].

4.2 Land use impacts after the actual occupation

As in most other impact categories, land use impacts do not occur only while land is being used, but may extend after the studied land use. Many assumptions must be made in order to assess the most likely future events, which are at all rates highly speculative. Two main approaches to estimate the future impacts may be used:

If land was abandoned at the end of the studied system (e.g. t_{fin} in Fig. 3, or t_2 in Fig. 4), land quality would probably continue evolving and might converge or not to a similar steady state than the dynamic reference situation. The impacts after the occupation may thus be estimated in a similar way to those occurring during occupation, for a relevant period of time. It is here recommended to consider relaxation until a new steady state is reached (t_{100} in Fig. 4).

Fig. 4 and the explanation above consider the effects of a 'pulse' intervention. An alternative way to deal with the impacts in the future is to consider a 'continued' land use, i.e. assuming that the present occupation continues to infinite (i.e. a very long period) and assess in parallel the likely evolution under natural relaxation from t_1. Then the impacts are divided by the time considered in order to estimate the impact per year of land use (**Fig. 5**). If the relaxation rate does not depend on the initial land quality, both approaches should give similar results of land use impacts per year.

Particularly in cases of severe transformation processes it is possible that land recovery after the human interventions does not reach the dynamic reference situation within the

[13] Müller-Wenk (1998, p.25) gives examples of re-naturalisation times for biodiversity, which can be used to characterise the land use impacts when natural recovery is assumed to be the most likely dynamic reference situation.

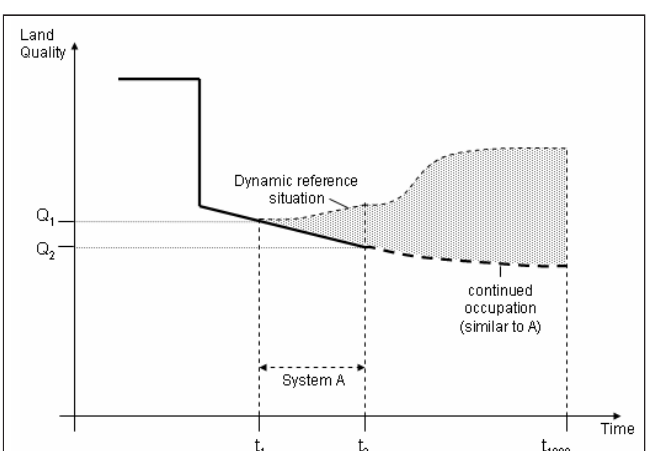

Fig. 5: Considering a continued occupation is an alternative way to assess the impacts after occupation. The shaded area is calculated over a long period of time and then divided by the calculation period (1000 years in the figure) to estimate the impact per year of occupying the land

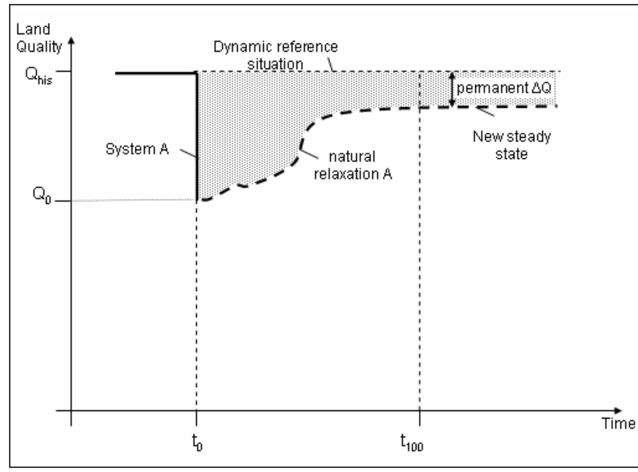

Fig. 6: After a transformation process, impacts are not reversed within the assessment time frame (t_{100})

scope of the impact assessment. In these cases the associated impact will grow indefinitely with the assessment time frame, giving an indication of the gravity of these irreversible impacts (**Fig. 6**). The new steady state reached with natural relaxation after the transformation process is not equivalent to the reference within the assessment time frame (in this case, t_{100}). In fact, the new steady state might represent a completely different quality (e.g. when an old quarry is filled with water, a pond is left where there was a meadow). These permanent changes in quality can be expressed with a qualitative note of 'permanent change of land cover', which needs to be interpreted on a case specific basis. Alternatively, the calculation of the impact size might also be done under the simplifying assumption that the relaxation could lead to the dynamic reference situation but only after a virtual relaxation time that is clearly longer than any real relaxation time, for instance 10,000 years.

The time frame of the assessment determines to a big extent the effects considered for land use. In LCIA in general, the total environmental impacts caused by the environmental interventions are the focus of attention (Jolliet 2005).

Depending on the impact pathway being assessed, it may take between some decades and several centuries to reach a new steady state, at which point it is likely that both the assessed and the reference system would be in an equivalent land quality. When this happens (e.g. around point t_{100} in Fig. 4) it is not relevant to have a longer time frame because all the impacts caused by the studied system have already been assessed. In case the impacts caused by the system are not fully reversed (as in Fig. 6) a qualitative note that irreversible impacts are detected at the end of the modelling time frame needs to be made for the interpretation phase. Consistently with the Life Cycle Initiative LCIA Programme, it is here suggested that characterisation factors for land use impacts should be calculated for two time horizons (Jolliet 2005):

- Overall impacts (baseline) over an infinite or very long term, at least until a new steady state is reached both for the reference and the studied system;
- 100 years as a shorter term with likely smaller uncertainties.

4.3 The need for bio-geographical differentiation[14] in land use impact assessment

There is a growing body of evidence that the site-dependency of environmental impacts needs to be incorporated in the normal LCA practice if we are to provide meaningful results (see e.g. Potting et al. 1998, Wenzel 1998, Huijbregts et al. 2000, Krewitt et al. 2001, Ross and Evans 2002, Frischknecht et al. 2004, Finnveden and Nilsson 2005). Land use impacts are highly dependent on the conditions of the place where they occur; i.e. the same intervention may have different consequences depending on the sensitivity and inherent land quality of the environment where it occurs. The main bio-geographical parameter determining land quality is climate (temperature and precipitation), with soil type, steepness, vegetation cover and (history of) land use type playing a relevant role as well. Other spatial-dependent impact categories (e.g. acidification; eutrophication; toxicity; etc.) attempt to solve this issue by defining dose-response functions dependent on the media receiving the relevant emissions. Two options to address the relevance of land use impacts are discussed here; it is a future challenge for LCIA to ensure a consistent application of the dose-response principle amongst the different impact categories.

The historical natural state or the potential (secondary climax) quality, derived from suitable surrounding ecosystems, may be used as context reference values for land use impacts. Milà i Canals et al. (2006) suggest using the 'distance to climax', defined as the difference between current land quality and the potential climax, as a contextualising factor to take the significance of land use impacts into account.

An alternative way to deal with the state of the studied land use relative to its context is the definition of thresholds, whereby the activities occurring either above or below critical thresholds are each treated accordingly. When a critical threshold is reached, a qualitative statement should be made to warn that any further use of that piece of land may lead to irreversible impacts. The definition and implementation is likely to be different for the different impact pathways: from a biodiversity point of view it may be preferred to use areas that are already degraded in order to preserve the ones with highest quality, whereas from a soil quality perspective it is often advocated to avoid using the most degraded areas in order to give them the time to be naturally regenerated.

Inventory information for bio-geographical differentiation in land use. Bio-geographical differentiation of land use impacts can possibly be done by geo-referencing the land use interventions (transformation and occupation) and linking this to global maps with climate patterns, vegetation and soil types, etc. Alternatively, the land use interventions may be defined within the LCI with a string of parameters or archetypes for 'situation differentiation' (Jolliet 2005) (e.g. 'land use x, continental central Europe, acid soil, mixed forest'). The practical details of this are out of the scope of this paper.

[14] In the literature the expression 'site-dependent' is more commonly found to refer to this concept (e.g. Udo de Haes et al. 2002). However, the term 'site' suggests a level of accuracy that is not necessary.

Currently available LCI databases do not contain full information on the geographical location of processes. For reasons of practicality, it is therefore necessary to propose default procedures for land use impact assessment in case of insufficient location information in LCI.

Bio-geographical differentiation in land use impact modelling. Additionally, different degradation processes may be responsible for the main effects on a certain impact pathway in different regions of the globe, which requires a spatial-dependent impact pathway modelling at the LCIA level. E.g.: erosion may be the main cause of biotic production potential degradation in certain regions of Africa, whereas salinisation may be more relevant in some regions of Australia. Even though the damage indicators for these impacts should be the same, the choice of midpoint indicators might be different (e.g. % of soil lost vs. electric conductivity or % exchangeable sodium). A way to interpret or combine these midpoint indicators should thus be made available in methods for land use LCIA.

5 Conclusions and Recommendations

Land use impacts should be assessed in LCA in order to keep the credibility of the tool particularly in land demanding sectors such as agriculture, forestry, mining, fishery, house-building or industry. The use of seabed requests particular considerations, which are briefly commented in **Box 1**. The most relevant impact pathways requiring protection within LCIA identified in this paper affect the natural environment (the existence value of biodiversity and ecological soil quality) and the natural resource aspect of soil quality and biodiversity (referred to as biotic production potential).

The selected impact pathways should be considered by future land use impact assessment methods, linking the land use elementary flows and modelling aspects registered in LCI to the impact pathways biodiversity; biotic production potential; and ecological soil quality. These impact pathways could eventually be linked to damages on the natural environment and resources (damage approach). The impact pathway 'ecological soil quality' could eventually lead to effects modelled in other impact pathways, such as global warming (effects on the carbon cycle) or toxicity (effects on the soil's filter and buffer capacity). For the moment, enough information exists to include at least the effects on the carbon cycle. In the case of biotic production potential, at least the effects on soil quality should be included, if not enough information on the effects of land use on useful species is available. The inventory needs to record information on the type of land use, its coverage (area) and intensity (measured in different parameters for each impact pathway), and the bio-geographical conditions of where the land use occurs. Both the effects of land transformations (when land characteristics are changed on purpose to accommodate a new land use) and occupations (when land characteristics are kept more or less constant in order to maintain a specific land use) need to be covered by land use impact assessment methods. Including the occupation impacts acknowledges that humans are not the sole users of land. The time lag between the land use intervention and

Box 1: Considerations on the impact assessment of seabed use

A special case of land use is the use of the seabed for different natural and human-controlled purposes, however the discussion group did not have enough expertise on the subject to properly address this issue, and so this box is merely intended to foster the debate. Ziegler et al. (2003) made a first attempt to include the damage of fishing to the marine natural environment in LCA. Many other sectors use the marine environment (fishery; marine mining; dumping of waste; shipping; etc.), affecting it in many ways. As a first classification, chemical, physical and biological effects may be considered. Chemical effects (e.g. water pollution with anti-fouling agents) should be included in ecotoxicity impacts. Biological effects of fisheries (direct mortality and discards) should be dealt with under depletion of biotic resources. Finally, the physical quality of the seabed is affected by penetration of e.g. fishing gears into the seabed (e.g. demersal trawls) and by the alteration of its characteristics, which indirectly influences biodiversity. The physical changes in the seabed are the land use related impacts.

The effect of fishing gears may be considered as an 'unintended transformation' of the seabed, or as a pulse occupation introducing changes in the seabed characteristics. Other human uses may be more similar to a 'pure' occupation (e.g. anchoring an oil platform or a windmill on the seabed) or a 'pure' transformation (e.g. extraction of substrate for sand mining). In any case, the human-induced effects will have to be assessed respect a reference situation without the human intervention. Tyler-Walters et al. (2003) selected three environmental factors caused by human activities, namely substratum loss (i.e. removal of the substratum), smothering and physical disturbance and abrasion. Changes in these factors are indirectly linked with biodiversity through the alteration of biotopes and habitats, which affects the occurrence and the abundance of species and communities.

Different marine landscapes have different sensitivities to human disturbance (Gubbay & Knapman 1999, Jones et al. 2000, Tyler-Walters et al. 2003), stressing the importance of bio-geographical differentiation also for the marine environment. Intolerance to substratum loss for instance is likely to be high but recovery may be rapid (< 5 years) in many sediment communities but will be much slower where long-lived, slow growing species are recorded.

As a starting point, it is suggested to focus the development of impact characterisation factors on the physical impacts occurring in the continental shelf, where it is likely that human impacts will affect most the seabed, and where more knowledge is available to derive these factors. These factors should link the physical changes introduced by human activities on the seabed to the effects on biodiversity, both from a natural environment and from a resource perspective.

the damage may be large, and the impacts on land quality should be assessed at least until a new steady state in land quality is reached by natural or human-induced relaxation. This new steady state may represent a permanent change in land quality respect the reference, which should be expressed in qualitative terms for a proper interpretation by the LCA commissioner. The size of impact is the difference between the effect on land quality from the studied land use and a suitable reference land use on the same area. The default reference land use is defined as the 'no use' of the same piece of land (i.e. natural relaxation), although alternative land uses may also be considered depending on the goal and scope of the study.

Land use impact assessment requires a proper *bio-geographical differentiation*. The level of detail of this differentiation depends on the goal and scope of the study:

Situation differentiation (Jolliet 2005). We recommend using this approach for land use impacts whenever the LCA user may not directly decide on the land management practices (e.g. food from a consumer and not from a farmer perspective; rock from a builder and not from a miner perspective; etc.). This may possibly be covered with a restricted set of characterisation factors for archetypical situations. The type of parameters defining the archetypical land uses include:

- Type of land use (mining; agriculture; forestry; uses leading to sealed soil; pasture; landfills; etc.) as well as its coverage and intensity
- Bio-geographical conditions to derive the likely surrounding ecosystems

When developing default values for the archetypical situations, values will have to be derived for 'unknown' types of land used, to be used when the conditions of the used land are unknown. In order to avoid the risk of undervaluing fragile areas, we suggest to base the default impact assessment values for unknown situations on worst-case conditions; consequently, if the results do not show relevant land use impacts the user may safely disregard them, whereas if the result is relevant the user would be motivated to look for more specific data on where the land use takes place.

Spatial differentiation. In the case of LCA aimed at providing information for land managers (e.g. agro-forestry LCA to detect environmental hotspots and suggest improvement opportunities; LCA of road construction; site-specific environmental product declarations of power generation; etc.), a more detailed *spatial differentiation* (Jolliet 2005) is required. The objective is to model the effects of different land management operations on the parameters influencing the impact pathways. Typically, the inventory should include the following information:

- Characteristics of the land use (defined to the required level of detail to allow the LCI modelling for the selected indicators)
- Bio-geographical conditions of the used land

The specific implementation of such approaches depends on the operational indicators chosen for the different impact pathways. The elements in both types of impact assessment are the same. The main difference will be that in the first case (situation differentiation) the assessment will have been done **for** the LCA practitioner, while a more specific (spatially differentiated) assessment will be done **by** the practitioner in 'land management LCA' studies.

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