**"Key elements in a framework for land use impact assessment within LCA"**

**Milà i Canals L[1*] (ed.), Bauer C[2], Depestele J[3], Dubreuil A[4], Freiermuth Knuchel R[5], Gaillard G[5], Michelsen O[6], Müller-Wenk R[7], Rydgren B[8]**

[1] Centre for Environmental Strategy, University of Surrey (D3), GU2 7XH Guildford (Surrey), UK
[2] Forschungszentrum Karlsruhe, Department for Technology-Induced Material Flows (ITC-ZTS), Hermann-von-Helmholtz-Platz 1, DE-76344 Eggenstein-Leopoldshafen, Germany
[3] Agricultural Research Centre - Sea Fisheries Department, Ankerstraat 1, B-8400 Oostende, Belgium
[4] Natural Resources Canada, 555 Booth Street, K1A 0G1 Ottawa, Ontario, Canada
[5] Agroscope FAL Reckenholz, Swiss Federal Research Station for Agroecology and Agriculture, Reckenholzstr. 191, CH-8046 Zürich, Switzerland
[6] Department of Industrial Economics and Technology Management, Norwegian University of Science and Technology (NTNU), NO-7491 Trondheim, Norway
[7] Institut für Wirtschaft und Ökologie, Universität St. Gallen, Tigerbergstrasse 2, CH-9000 St. Gallen, Switzerland
[8] SwedPower AB, Box 1842, SE-581 17 Linköping, Sweden
[*] Corresponding author (L.MiC@surrey.ac.uk)

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## Abstract

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, key elements in a Life Cycle Impact Assessment (LCIA) framework of land use have now been treated and are presented in this paper.

This framework 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). 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, thus requiring a proper level of bio-geographical differentiation. The time lag between the land use intervention and the impact may be large, implying that the 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.

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. Bio-geographical differentiation is required for land use impacts. For the moment, an indication of how such task could be done and likely bio-geographical parameters to be considered are suggested. The definition of suitable indicators for the suggested impact categories is a matter of future research.

## Keywords

Land use; land use impacts; LCIA; biodiversity; soil quality; dynamic reference situation; natural environment; natural resources; Bio-geographical differentiation; land quality; LCA

# 1. 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. 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.

LCA was developed as a site- and time-independent environmental assessment tool for (industrial) product development, while other impact assessment tools (mainly Environmental Impact Assessment, EIA) were developed to assess the impacts of localising a project in a specific site. The main differences between the two assessment methods can be stated with the following table:

| LCA | EIA |
|---|---|
| Assesses impact in relation to functional unit of a product or (industrial) process. This makes it a strong tool for analysis of alternative ways of producing the same economic good. | Assesses absolute (or total) impact of an activity, project or similar. This makes it a strong tool for site-specific management. |
| Not suited to be site-specific, but possibly site-dependent. Spatial scale is not well internalised into the method. | Is almost always site-specific when correctly applied. EA methodologies which can be used for non-site specific assessment do exist, e.g. SEA. Spatial scale is well internalised into the method. |
| Is stronger on global impacts than local. The larger the scale, the bigger the problem of LCA to account for the spatial variability of end-point impact and natural resource *quality*. This is particularly true for land use impacts. | Is stronger on local impacts than on global. There is no inherent problem in using a specially developed characterisation tool (*quantitative and/or qualitative*) since there is no need for comparisons beyond the decision-making unit (e.g. municipality, district). |
| Assesses impacts for a defined time span. Everything that happens beyond that time is treated as an outflow from the analysed system, and not quantified further. | In theory assesses impacts over infinite time, but in reality introduces a time-dependency through a differentiation between reversible and irreversible damages. However, the reversibility is rarely defined in terms of time. |
| Can be utilised for the development of input data into an economic/financial analysis. | Does not easily generate data which are quantifiable in economic/financial terms. |
| Toolbox with rather LCA-specific methods. ISO-standardised, thus rather stringent. Very difficult to penetrate for a lay person. | Framework which is open to the inclusion of many analytical tools. Applications vary considerably, and guidelines are not stringent. Relatively easy to penetrate for a lay person. |

From the above it may seem quite simple to identify the differences between EIA and LCA, but when LCA moves into land-use-related issues, and also increasingly towards spatial-dependent assessments, the clear borders between the two become somewhat less distinct. However, it is important to realise that for some types of decisions EIA is a more adequate tool than LCA.

[1] http://lcinitiative.unep.fr/. Taskforce on Resources and Land Use within the UNEP-SETAC Life Cycle Initiative Working Group on LCIA, hereafter called TF2. See http://www.lci-network.de/lciacorner.

## *1.1. Why is it important to assess land use impacts?*

According to UNEP (2002), the main driving force leading to pressure on land has been increasing demand for food production. The main human activities contributing to land degradation are highlighted in Table 1.

Table 1: Extent and causes of land degradation (UNEP 2002).

| *Degradation extent* | *Cause* |
|---|---|
| 680 million ha | **Overgrazing** — about 20 per cent of the world's pasture and rangelands have been damaged. Recent losses have been most severe in Africa and Asia. |
| 580 million ha | **Deforestation** — vast reserves of forests have been degraded by large-scale logging and clearance for farm and urban use. More than 220 million ha of tropical forests were destroyed during 1975–90, mainly for food production. |
| 550 million ha | **Agricultural mismanagement** — water erosion causes soil losses estimated at 25 000 million tonnes annually. Soil salinization and waterlogging due to poor irrigation practices affect about 40 million ha of land globally. Improper crop rotation and frequent use of heavy machinery are cited as well. |
| 137 million ha | **Fuelwood consumption** — about 1 730 million $m^3$ of fuelwood are harvested annually from forests and plantations. Woodfuel is the primary source of energy in many developing regions. |
| 19.5 million ha | **Industry and urbanization** — urban growth, road construction, mining and industry are major factors in land degradation in different regions. Valuable agricultural land is often lost. |

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). The need for their assessment 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], 1979 Convention on the Conservation of Migratory Species of Wild Animals[4], Ramsar[5], etc.).

There has been extensive debate on land use impact assessment in LCA, including two SETAC working groups (Udo de Haes *et al*. 1996, Lindeijer *et al*. 2002); a special issue in the Journal of Cleaner Production (issue 8: 2000, see Lindeijer 2000a); and extensive literature including the main references in LCA development (Heijungs *et al*. 1992, Fava *et al*. 1993, Audsley et al. 1997, Cowell 1998, Mattsson *et al*. 1998, Müller-Wenk 1998, Blonk et al. 1997, Lindeijer *et al*. 1998, Baitz *et al*. 1999, Goedkoop & Spriensma *et al*. 1999, Köllner 2000, Lindeijer 2000b, Guinée *et al*. 2001, 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). The present paper is the contribution from the UNEP/SETAC Life Cycle Initiative to the international debate.

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 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. These value judgements include the following:

- What are the functions of land that need protection? (section 2)
- Perception of "ownership": is land for human use or do we have to share it with other users? (occupation and transformation impacts; see section 3)

[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

- 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? (section 5.1)
- 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 5.2)
- Perception of land's recovery capacity: is land robust or fragile? See e.g. discussion on thresholds in section 5.3.

## 1.2. Land use impact assessment in LCA studies

From an LCA perspective, land use is an 'environmental intervention', consisting of human activities aiming at converting or at maintaining the state of a given land area (including soil, vegetation cover and artificial structures) in such way that it is suitable for an intended human purpose. In the Life Cycle Inventory (LCI) phase, such land use can be described by the type of land use (e.g. forest land; cropland; grassland; wetland; settlement; etc.), the land management activities (e.g. type of fertiliser use; % of surface sealed; etc.), the area size and its location (either geographically referenced or described by its climate region: tropical; temperate; etc.), and the duration of the use. The level of detail by which the intervention 'land use' should be described in the LCI depends on the goal of the study and the nature (intensity) of the land use activities.

The LCIA phase is aimed at determining the damaging or beneficial effects that originate from the use of land to finally arrive at the representative environmental damages (often referred to as 'endpoints' or 'areas of protection', Jolliet *et al*. 2004). Impacts arising from the use of land include a wide range of environmental issues. However, for the purposes of this paper, we refer only to the impacts on land quality itself that are not consistently included in current LCA practice: impacts on biodiversity[6] and soil quality. The inclusion of these impacts in LCIA is crucial since the production of raw materials (fibres, food, energy carriers, metals etc.) often takes place in ecologically fragile areas. These two main impact pathways are referred to generically as "land use impacts", and are discussed in section 2.

Land use also affects many other impact categories, such as climate change (through the role of land as a source or a sink of carbon); eutrophication and acidification (through e.g. removal of standing biomass); toxicity (through application of pesticides); etc. These impacts are currently addressed in the relevant impact categories, and will not be the subject of discussion in the present paper.

[6] 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).

# 2. Description of impacts from land use

Land provides the support functions for life, including cycling of nutrients, water and carbon, and the provision of habitat for both human and non-human life. These functions are paramount both for ecosystems and for humans (UNEP 2002), as they represent a key factor for agriculture and forestry, as well as for using land as a sink for waste. Other authors (Teller *et al*. 1995; Lindeijer *et al*. 2002; Candinas *et al*. 2002; Milà i Canals 2003, p. 182) address these aspects when listing the main functions of land. 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 (and should be included in the LCA through the functional unit). On the other hand, the environmental (sometimes called ecological) functions of land, most often externalised from economic assessments, should be covered in a method addressing land use impact assessment. For the purposes of this paper, we refer only to the impacts on land quality[7] itself that are not consistently included in current LCA practice:

- impacts on biodiversity[8], seen as a key element of the biotic natural environment (existence value of wild species), as well as a natural resource for its provision of valuable species for humans (use value of wild species),
- and soil quality or the long-term capacity of soil to sustain life support functions, which are a key element of the semi-biotic[9] natural resources exploited by humans (biotic production) as well as of the natural environment.

At this point, it is important to stress that, as in the case of freshwater, both humans and non-humans benefit from land's functions, which is a major difference between land and freshwater and other resources[10]. This list of damage categories affected by land use highlight the dual nature of land as a resource and as a habitat, which has consequences on the types of impacts that need to be covered by the framework for land use impacts assessment[11]. In principle all types of land (pristine, transformed –e.g. agricultural-, coastal, deserted, etc.) need to be covered by any land use impact assessment method because some species will be using some of its functions somewhere.

It is not excluded here that further effects will have to be considered in LCIA. Indeed, land use may have an important influence on the global climate system (mainly due to its role as carbon sink or source[12]), as well as on nutrient emissions leading to eutrophication and acidification; release of toxic substances; etc. However, these effects need to be considered in the modelling of the relevant impact categories.

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[7] 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.

[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] In general, all natural resources stored in the biosphere might be essential for non-humans (e.g. copper in soil is an essential nutrient). However, these other resources are generally included in the aspect "soil quality".

[11] Freshwater also presents a dual nature resource-habitat, and a similar framework might be applicable to its impact assessment.

[12] Carbon stored in soil as organic matter is an important pool in the global carbon cycle (with an estimated 1400-1500 Pg C according to Post *et al*. 1982 and Eswaran *et al*. 1993, compared to 750 Pg C in the atmosphere, 550 in vegetation, or 5000 in fossil fuels, Brady and Weil 1999, p. 447). 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. 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 2004; Svensson 2005).

Jolliet *et al*. (2004) argue that arable land might be considered as man-made, and therefore the changes on the human-induced land properties be considered as a damage to the man-made environment. However, including man-made environment as a damage in LCIA is still the matter of debate, as it is possibly more relevant to assess human activities as part of the studied system rather than as entities being impacted. Therefore, this TF2 suggests that soil (including arable soil) should be considered as a natural resource in its main part, even though it is shaped and modified by humans. Consequently, the modelling up to the damage on man-made environment is not pursued further in this paper.

The following sections describe these impact pathways in a qualitative way, from the LCI interventions to the damage categories. Each intermediate point of these pathways is eligible as midpoint indicator, and is up to the LCIA method developer to choose one or another. For the damage indicators, an effort has been made to follow the recommendations by Jolliet *et al*. (2004) whenever possible. In order to choose among midpoint indicators (which is out of the scope of the present paper), the criteria suggested by the TF1 in the LCIA Programme of the Life Cycle Initiative[13] should be brought in mind, as well as the recommendations of the ISO standard on Life Cycle Impact Assessment (ISO 2000).

## *2.1. Effects of land use on biodiversity: impacts on the biotic natural environment and on biotic resources*

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.
The changes in species composition affect both those species without a direct use by humans (this effect must be considered because of the existence value of these species) and those which are useful for humans (i.e. with a use value). The latter are particularly relevant in agricultural systems (Swift and Anderson 1994), e.g. for pollination and biological pest control.

## *2.2. Effects of land use on soil quality: impacts on semi-biotic natural resources and environment*

Soil quality is the result of the interaction in time of biosphere and lithosphere. Therefore, when a certain land cover is transformed to make place for another type of activity (e.g. a forest is cleared to make space for agriculture; a plot of grassland is partly sealed for urbanisation; etc.), big effects can be expected on the quality of soil at the level of environmental impacts. Also when a certain land use is maintained over a period of time, the activities performed on land may continue to impair or enhance soil quality, and the fact that soil quality is maintained at a different level due to the existence of the studied system needs to be included as well. It must be noted here that the definition of soil quality is multi-faceted, and when one aspect of soil quality is improving another might be declining. 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. The rest of life support functions, albeit crucial for human welfare,

[13] http://www.lci-network.de/cms/content/pid/568

are not directly used by humans and it is probably better to assess them separately as part of the natural environment.

### 2.2.1. The natural resource aspect of soil quality

A natural resource is a part of nature that is used as an input in a human-controlled process delivering goods demanded by human society. Typically, the resource is degraded during this process, which means that the total availability of the resource on the globe gets reduced. The current annual reduction is negligible for amply available resources and bigger for other resources. This availability reduction may be prevented by counteractive processes (recycling, substitution) which generally require an extra effort.
According to Milà i Canals *et al*. (submitted) 'fertile land, as defined by its soil (chemical, physical and biological quality) and the properties of its geographical location (slope, availability of water, radiation, temperature, etc.), is possibly the most relevant type of land from a resource scarcity perspective'. Fertile land is used every year again as an input into man-controlled food, fibre and fuel production processes. Unfortunately, this 'flow-type' resource may be deteriorated by certain land 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 (at least during the occupation).

The spatial aspect of land as a resource (i.e. surface to build human habitat and infrastructures) is seen more of an economic issue (competition between human uses; Lindeijer *et al*. 2002). It is suggested to treat this as part of the economic system, and not of the impact assessment. Obviously, the environmental impacts derived from such use (e.g. on biodiversity; on the soil quality due to surface sealing; etc.) need to be assessed in the LCIA.

### 2.2.2. The natural environment aspect of soil quality

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 and nutrients cycles, as a filter and buffer of hazardous chemicals, and as a habitat of soil flora and fauna 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; the increased or decreased levels of soil nutrients caused by agriculture affect soil organisms (apart from the off-site effects through nutrient emissions, already considered in the relevant impact categories such as eutrophication, acidification and global warming).

# 3. 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 Figure 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 Figure 2-3). If land use processes are listed in life-cycle inventories LCI, it is usual to call them interventions, similar to all other types of LCI entries. Occupation interventions are measured in surface-time units (e.g. ha yr), representing a certain type and amount of land (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). In contrast to the simplified assumption that occupation and transformation are separate processes, the environmental quality of a land area may gradually change also during an occupation process (see Lindeijer *et al*. 2002, p. 42-44).

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 (see Figure 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 and on the site's bio-geographical conditions. The transformation impact is represented by the shaded area between the dotted line and the full line in Figure 1, i.e. the integral of $\Delta Q$ over time. The magnitude of this impact may be coarsely approximated by a triangle. The triangle is large if the recovery is slow due to the depth (intensity) of transformation and/or adverse bio-geographical conditions. The figure assumes that land quality would have not changed without the studied system, and so the reference situation (section 5.1) is the initial state of land quality.

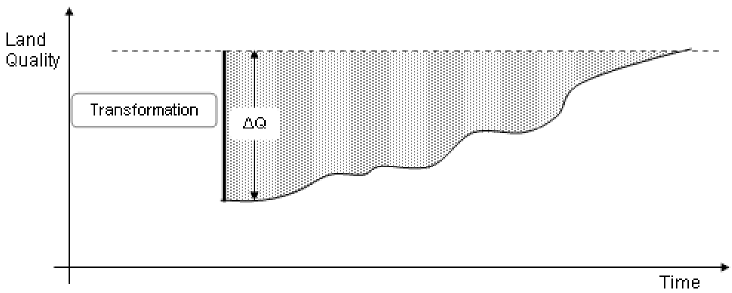


Figure 1: Pure case of land use by a transformation process not followed by any occupation process. $\Delta Q$ represents the change in land quality and the shaded area represents the occupation impact.

It must be noted in Figure 1 that land quality may actually be represented by different parameters, depending on the specific impact pathway (e.g. biodiversity; 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 (Figure 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.

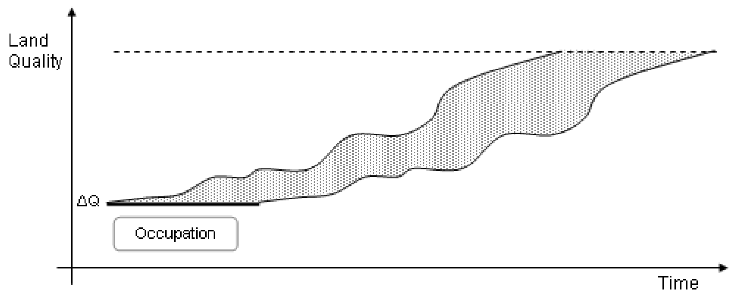


Figure 2: Pure case of land use by an occupation process with no land quality change during occupation (ΔQ=0). The original transformation process was somewhere in the past. The shaded area represents the occupation impact.

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 small transformation processes, causing intended or non-intended changes of land quality. Sometimes the transformation is dominant (e.g. when a forest is transformed into a residential area) and sometimes the occupation is (e.g. once the residential area is established, its use as residential area). The impact can be combined as a mix of transformation impact and of occupation impact: additivity of transformation impact and occupation impact is given because both terms represent a land quality difference during a certain time. Land uses may thus be defined in terms of their *coverage* (quantity of area used) and *intensity* (degree and symbol of quality change). Considering a mixed situation between Fig. 1 and 2, Table 2 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.

Table 2: 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 (e.g. soil quality) while decreasing for others (e.g. biodiversity). |
| **$t_0$ 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 (going up and down or constant level), 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 a human induced transformation process, 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. |

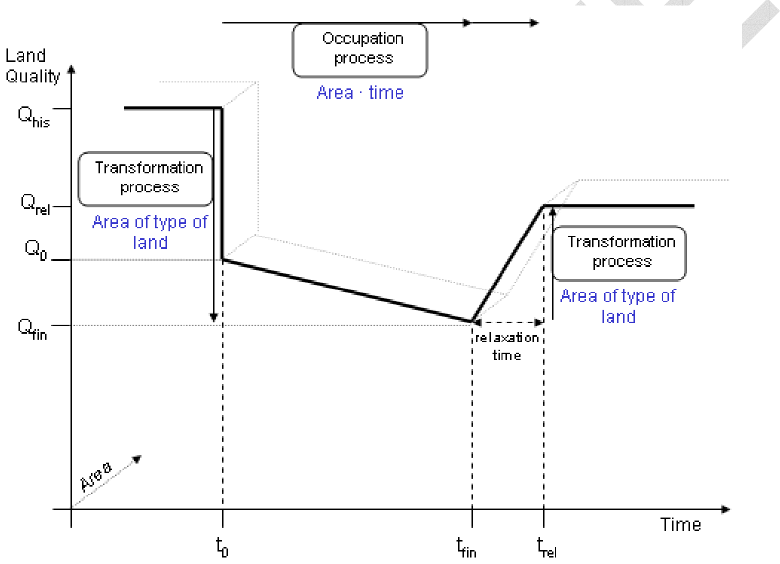


Fig. 3: Evolution of land quality with land use interventions (adapted from Lindeijer *et al*. 2002).

The changing land quality between $t_0$ and $t_{rel}$ 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 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 (occupied surface)
- Time (duration of the occupation + 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

# 4. Indicators for land quality

In the LCIA, impacts on the relevant impact categories (at the midpoint or endpoint 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 (at least natural environment and natural resources). The impact indicators for each of these impact pathways require information on different parameters from the LCI in order to be calculated.

In the case of natural environment, possible indicators to explore are those used by UNEP-WCMC[14] as pressure (for transformation interventions) and state (for occupation) indicators for plants and animals (Jolliet *et al.* 2004). The effects of land use on biodiversity relate to the damages on the biotic natural environment, for which no damage indicator has yet been agreed (Jolliet *et al*. 2004). However, it may be fruitful to try to work with PDF (Potentially Disappeared Fraction of species) or PAF (Potentially Affected Fraction of species), as these indicators are currently used by eco-toxicity categories (Jolliet *et al.* 2004), and might be related as well to impacts from eutrophication and/or acidification. One problem with these indicators is that they are only quantitative and do not deal with qualitative aspects such as relative scarcity of species; see Kylakörpi *et al.* 2005 for a method combining quantitative and qualitative biodiversity information.

At the level of midpoint indicators for the natural environment aspect of soil quality, some authors try to assess the effects of land use on several indicators for life support functions: pore volume; heavy metals content; soil organic matter; nutrients levels; soil biota; rooting depth; etc. (e.g. Cowell 1998; Mattsson *et al*. 1998; Baitz *et al*. 1999; Oberholzer et al. 2006). The problem of such an approach is that these indicators require further aggregation in order to be useful for the overall LCA results. As an operational simplification, Milà i Canals (2003) suggests the use of soil organic matter as a single indicator representing all life support functions of soil and soil resilience.

In the case of natural resources, counteraction to compensate soil quality deterioration (soil recycling, substitution of soil) appears to be possible but involves extra efforts. In analogy to fossil resources, this extra effort (expressed in energy, environmental or other suitable units) may be taken as a damage indicator of the impact on resource availability caused by the resource use. If this is done, this would contribute to better interpretable LCIA results, because most natural resources are then treated alike in LCIA. In any case, the indicators used should possibly cover not only the 'surplus energy' but also other emissions related to the restoration activities, leading to other damages. This is because recent research suggests that the impacts from land use-based activities are not properly represented by energy indicators (Walk *et al*. 2005; Huijbregts *et al*. 2005).

At the midpoint indicators level, the production of biomass (Lindeijer *et al*. 1998; Weidema and Lindeijer 2001) or yield gap indicators (Bindraban *et al*. 2000) have often been suggested for soil fertility. However, production-based indicators have been criticised because low-quality soils may have high yields because of many other factors, chiefly skilled agricultural management (Burger and Kelting 1999; Bouma 2002). Indeed, yield-based indicators are better suited for assessing economic than environmental consequences of land management. The latter approach may be further simplified to the assessment of the most important soil degradation process (surface sealing; erosion; salinisation; loss of soil organic matter; compaction; build-up of toxic substances; etc.) or "limiting factor" for biomass production in each land use case (e.g. Feitz and Lundie 2002; Milà i Canals *et al.* submitted). This represents a similar approach to the site-dependent impact

[14] World Conservation Monitoring Centre.

assessment of eutrophication, where different sets of characterisation factors may be derived depending on whether the limiting nutrient in the receiving ecosystems is P or N.

Extensive work still needs to be done for the definition of land use impact indicators, which is the focus of ongoing work within TF2 and a forthcoming expert workshop[15]. Table 3 is a non-exhaustive list of potentially useful indicators for land use impact pathways, with likely requirements in terms of LCI parameters:

Table 3: Examples of impact pathways indicators for land use impacts and requirements of LCI information.

| ***Impact pathway*** | ***Indicator*** | ***Level*** | ***LCI modelling aspects*** |
|---|---|---|---|
| Natural environment - biodiversity | 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 |
| Natural environment - biodiversity | % 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) |
| Natural environment - biodiversity | Red-listed species; key features | Midpoint | Species correlation with habitat, the ecological habitats found and affected area (Kylakörpi *et al.* 2005) |
| Natural environment - biodiversity | 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) |
| Natural resources - fertile land | Surplus energy (+ possibly other interventions) [c] | Damage | Requirements to restore soil quality (to a level considered "equivalent" to "before the land use" by experts or legislation) 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.* submitted) |
| Natural resources - fertile land | Deficit of Soil Organic Matter (SOM) [Mg SOM year] | Midpoint | Changes in SOM due to the studied system, which may be measured/ calculated/ estimated by different means (Milà i Canals *et al.* submitted); additions of organic matter (e.g. manure; crop residues); effects of agricultural practices on degradation rates |
| Natural resources - fertile land | 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 |
| Natural environment - soil quality | PDF or PAF | Damage | To be explored, with the limitations discussed above |
| Natural environment - soil quality | 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).

[15] http://www.soc.surrey.ac.uk/ias/workshops/DEFNBEST/cfp.php

# 5. Proposal for a way forward: Considerations for the application of the framework

This section deals with how to assess the elements required to include land use impacts in LCA; the discussion is organised in the following sections:

- Section 5.1 defines a suitable reference situation to assess land use impacts in LCIA.
- Section 5.2 deals with how to include the impacts from land use occurring after the system's temporal boundary within LCIA, including the time frame over which the above mentioned impacts need to be assessed.
- Section 5.3 discusses the need for bio-geographical differentiation in land use impact assessment, and
- Section 5.4 suggests how to cope with this differentiation in practice.

## *5.1. 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. In LCA, the basic question to be answered is "Which are the additional damaging effects on nature, if the current activities of the economical system are increased by the production of N units of a given product, causing emissions and uses as tabled in the LCI of the corresponding production process chain?" If the production of 1000 glass bottles causes a $SO_2$ emission of 2 kg, the damaging effect of this additional emission of 2 kg is compared to the 'reference situation' *without* this additional emission, in order of finding the *net* effect of the production to be examined. Thus it is clear that the reference situation for assessing an emission of 2 kg $SO_2$ is the non-emission of these 2 kg $SO_2$.

If the LCI of the production to be examined contains an entry 'use of an area of size 1 $km^2$ for soybean production during 1 year', the reference situation is a bit less obvious than in the case of the $SO_2$ emission. It has been suggested to use either the historic natural land *state* (Blonk *et al*. 1997; Brentrup *et al*. 2002; $Q_{his}$ in Figure 1) or the potential *state* after relaxation (Lideijer *et al*. 2002; $Q_{rel}$ in Figure 1) as a reference *state*. These references have the advantage of giving information on the distance to the land's 'best (historic / potential) quality', and therefore the relevance of the impacts caused by the studied system. However, using a *static state* as a reference for occupation impacts does not take into account the dynamic nature of land evolution, and raises problems of allocation between successive land uses.

By analogy to the abovementioned $SO_2$ emission, the *dynamic reference situation* (baseline, in the Kyoto protocol terms) is here the *non*-use of the area of size 1 $km^2$ for soybean production during 1 year. But this raises further questions:

1. Does 'non-use' of the 1 $km^2$ area 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 1 $km^2$ area is not used for the studied human activity (soybean production), what are then the characteristics and quality evolution of this area (both in the case when it is totally free from any human land use and if it is occupied for any other activity)?

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; Ekvall *et al*. 2005), i.e. relative to a situation where this activity is not undertaken.

Consequently, the adequate reference situation for retrospective LCA studies is natural relaxation. This is shown in Figure 4, where the land use 'A' from $t_1$ to $t_2$ is assessed against the non-use of land for system 'A' (i.e. natural relaxation from $t_1$). Again, it needs to be stressed that Figure 4 is 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. As in most impact categories, land use impacts do not only occur during the actual interventions (land occupation and/or transformation), but there is a time lag in the occurrence of part of the impacts. Figure 4 only shows the impacts during the occupation process, and section 5.2 deals with the impacts occurring after the occupation process.

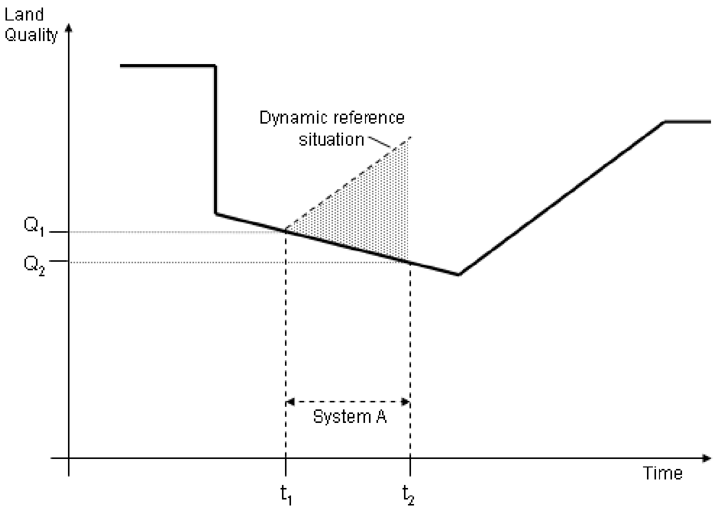


Figure 4: Impacts during the actual land use of system 'A' using natural relaxation as the dynamic reference situation (shown by the dashed area).

On the other hand, if the study aims at assessing the consequences of changes in the studied land use, e.g. conversion to alternative uses (prospective or consequential LCA[16]), only the changes in land use impacts directly due to the studied system respect an alternative system are considered. To follow with the analogy, in a consequential LCA not all $SO_2$ emissions are accounted, but only those that change due to the studied system. Therefore, when calculating the difference in land use impacts between the studied and the alternative system, the natural relaxation reference goes automatically off the calculations, and one of the systems becomes the reference[17]:

(Ref-A) – (Ref-B) = B-A

where A is the land quality evolution under system A; B is the land quality evolution under system B (alternative land use); and Ref is the original dynamic reference situation (natural relaxation). The dynamic reference situation must be defined in the goal and scope definition, consistently with

[16] Consequential LCA focuses on the effects of substitutions among alternative product systems (Weidema 2001), or the consequences of changes in the studied system (Ekvall *et al.* 2005).
[17] Milà i Canals *et al.* (submitted) suggest that in consequential LCA the reference should be defined as the 'most likely alternative situation if the system under study was not established there'. 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. In cases where the alternative land use implies a lower land quality than the studied land use, the system may be credited with the positive effects on land quality. It must be noted that natural relaxation may also be the reference situation in consequential LCA, whenever natural relaxation 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.* submitted).

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 characteristics and quality evolution of this land area when the studied activity is not established there, some expert judgement and modelling expertise will be required to assess the natural relaxation or any other alternative land use situation. In 'land management LCA' (e.g. focused on agriculture; forestry; mining; landfilling; etc) such expertise will probably be available to the LCA practitioner. 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 for the location of the land area is determined on the basis of global land cover maps, whereby the natural land cover of the location can be 'interpolated' from the grid cells in the neighbourhood being in natural or near-to-natural 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. This two steps procedure applies to the biodiversity impact pathway, but may be applicable also for the soil quality pathways. 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. This is further discussed in Section 5.4.

## 5.2. 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 Figure 3, or $t_2$ in Figure 4), land quality would probably continue evolving and might converge or not to a similar steady state than the dynamic reference situation (Figure 5). The impacts after the occupation may thus be estimated in a similar way to those occurring during occupation, for the period of time suggested by the LCIA methodology (see below). It is here recommended to consider relaxation until a new steady state is reached ($t_{100}$ in Fig. 5).

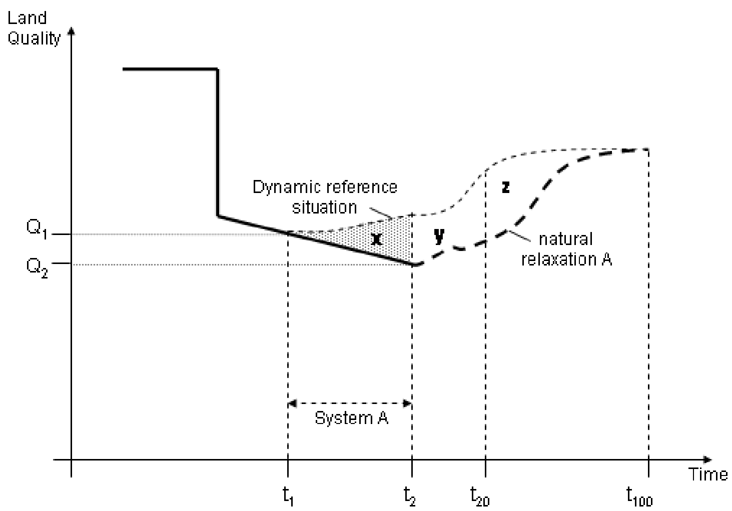


Figure 5: Land use impacts are calculated as well after the end of the actual occupation process ($t_2$). The impacts allocated to system A depend on the time frame of the impact assessment ($t_{20}$ and

$t_{100}$ 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.

Figure 5 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 (see Fig. 6). If the relaxation rate does not depend on the initial land quality, both approaches should give similar results of land use impacts per year.

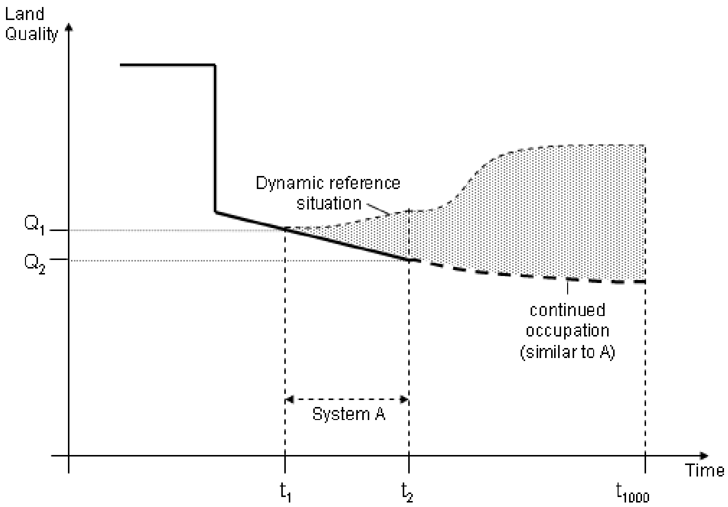


Figure 6: Considering a continued occupation is an alternative way to assess the impacts after occupation. The shaded area is calculated over a long period of time and then divided by the calculation period (1000 years in the figure) to estimate the impact per year of occupying the land.

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 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. In Figure 7, an important transformation process leads to a decrease in land quality; if the activity assessed was actually this transformation, the dynamic reference situation before the studied land use would be defined as the continuation of the previous land use (at $Q_{his}$). After the transformation process (which in the Figure has a very short duration, for simplicity), natural relaxation cannot reach the reference within the assessment time frame (in this case, $t_{100}$).

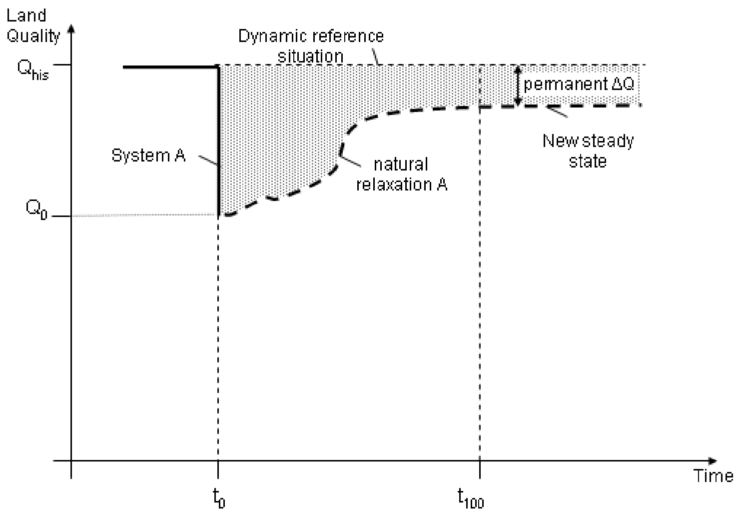


Figure 7: After a transformation process, impacts are not reversed within the assessment time frame ($t_{100}$).

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. I.e. all the effects over infinite time of a particular emission need to be captured by the impact assessment method, although shorter time frames (e.g. 'short-term': decades; 'mid-term': around one century; 'long-term': over 100 years) are usually considered for practicality and to reduce our uncertainty over future events (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 Figure 5) 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 Figure 7) it is relevant to have a longer time frame because the continuing degradation would be highlighted; however, the practitioner needs to be aware of the fact that modelling such a long time frame introduces big uncertainties in the results. A way to deal with this uncertainty is not modelling the impacts quantitatively over the whole time frame, but adding a qualitative note that irreversible impacts are detected at the end of the modelling time frame. 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.

## 5.3. The need for bio-geographical differentiation in land use impact assessment

Originally, LCA was conceived as a site-independent environmental assessment, mainly due to data availability and the nature of the assessment. Indeed, in industrial product development (the application for which LCA was conceived) mainly the technology type needs to be assessed with LCA, which advocates for a site-independent analysis: it is not practical or even possible to collect site-specific information for all sites included in the LCA (Finnveden and Nilsson 2005). Nevertheless, this may not hold true for some applications of LCA, and particularly for some impact categories such as land use impacts. Indeed, 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.
Wenzel (1998) discusses on the issue of site-dependency as related to the type of decision to be made based on the LCA results (application), concluding that the LCA applications needing to be site-dependent are mainly the assessment of Best Available Technologies, choice between alternative suppliers, and marketing. Ross & Evans (2002) maintain that excluding temporal and spatial site-dependent information to support decision making at the policy making level reduces the usefulness and credibility of LCA results. Finnveden and Nilsson (2005) mention regional planning as one application of LCA for which there is generally the required information for site-dependency, and where it is relevant for the LCIA to be site-dependent.
Other authors have pointed out that, regardless of the application of the results, site-dependency is needed for some impact categories (particularly those having effects at regional or even local levels). Some authors have concluded that including site-dependent data in the assessment results in a significant variation in the damage factors (Potting *et al.* 1998 for acidification; Huijbregts *et al.* 2000 for acidification and terrestrial eutrophication; Krewitt *et al.* 2001 for $SO_2$, $NO_x$, fine particles and NMVOC for impacts on human health, acidification, eutrophication and man-made environment; Finnveden and Nilsson 2005 for some impacts related to human health, where a sub-country level of detail in site-dependency might be relevant to consider).
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 (i.e. the species composition of a tropical forest is different to that of a temperate grassland or a public garden; the soil quality in a dry savannah is different to the soil quality in an Alpine meadow; etc.). The main bio-geographical parameter determining land quality[18] is climate (temperature and precipitation), with soil type, steepness, vegetation cover and (history of) land use type playing a relevant role as well. The impacts derived from an identical land use (in terms of economic activity) will therefore vary according to the place where the land use is based: the effect magnitude of a transformation forest-to-meadow on each of the impact pathways varies strongly between a hot/humid region and a cold/humid or cold/dry region.

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 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.* (submitted) suggest using the 'distance to 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 above critical thresholds are all treated in the same way. 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 (with low biodiversity) in order to preserve the ones with highest quality. On the other hand, in the case of soil quality it is

[18] It must be stressed that land quality may be defined in many different ways, depending on the interests of who defines it. Here we use this term in a general way, stressing that its measurement lies at the level of the different impact pathways.

often advocated to spare the most degraded areas in order to give them the time to be naturally regenerated, while the most fertile areas (e.g. 'prime farmlands'), which have a higher resilience and potential for biomass production, are to be used preferentially for food and fibre production.

### 5.3.1. Inventory information for site-dependency in land use

The site-dependency of land use impacts needs to be captured explicitly by LCA, possibly by including the relevant bio-geographical parameters within the LCI information and/or in the impact pathway modelling (LCIA). This can possibly be done by geo-referencing the interventions (transformation and occupation) and linking this to global maps with climate patterns, vegetation and soil types, etc. or by defining land use interventions within the LCI with a string of parameters or archetypes of "situation differentiation" (Jolliet 2005) (e.g. 'land use x, continental central Europe, acid soil, mixed forest'). The practical details of this are to be dealt with in the future paper on operative methods for land use impact assessment. This future paper will also have to consider the level of detail that can be achieved in the space information depending on the goal of the study; i.e. 'situation differentiation' should be possible with a limited amount of information (e.g. region or type of biotope) for generic assessments, while LCA focused on land management will require detailed 'spatial differentiation' (e.g. soil and vegetation type; climate; etc.) in order to determine the best management options.

### 5.3.2. Site-dependency in land use impact modelling

Additionally, different degradation processes may be responsible of the main effects on a certain impact pathway in different regions of the globe, which requires a site-dependent impact pathway modelling at the LCIA level. E.g.: erosion may be the main cause of soil quality 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 'early-midpoint' indicators might be different (e.g. % of soil lost vs. electric conductivity or % exchangeable sodium). A way to interpret or combine these 'early-midpoint' indicators should thus be made available in methods for land use LCIA.

## *5.4. Practical implementation of bio-geographical differentiation in LCIA*

It should be born in mind that LCA studies may cover a wide range of land uses, which may require different levels of detail in the assessment of land use impacts:

- In some occasions, the LCA study is focused on an end product (e.g. a coffee machine), which uses many components with related land use impacts; in these cases, only a generic assessment of the land use impacts is possible[19]. This may possibly be covered with a *situation differentiation* (Jolliet 2005) with a restricted set of characterisation factors for archetypical situations.
- On the other hand, there is growing interest in the application of LCA to the stages of product's life cycles where land management is crucial (e.g. agriculture; forestry; landfilling; mining; etc.). In this second case a specific assessment of the land use impacts is required, because the land use impacts may change enormously depending on the land management practices, which are one of the main subjects of the LCA study. *Spatial differentiation* (Jolliet 2005) is then required possibly at a very local level, in order to derive characterisation factors specific for the environmental conditions of the used land.

[19] Land use impacts may still dominate the environmental profiles of some 'end products' (e.g. bread; bio-diesel; pork chop; etc.); however, unless the assessment is done with the aim of affecting the land use stages it is not possible and probably not relevant to have a sophisticated land use impact assessment for this type of studies.

The elements in both types of impact assessment are the same. The main difference will be that in the first case the assessment will have been done for the LCA practitioner, while a more specific assessment will be done by the practitioner in ‘land management LCA’ studies. In the first case, the practitioner will possibly find the results in the form of characterisation factors associated with different archetypical land use types listed in the LCI (e.g. 1 $m^2$year of sustainably managed forest, Northern Europe; 1 $m^2$year of sealed industrial land, Korea; etc.). For LCA of land management activities, on the other hand, the practitioner may be interested in a more detailed analysis using more specific information from the LCI (e.g. restoration activities after a certain length of mining or landfill; tillage, fertiliser use, maintenance of margins, etc. in agricultural land; etc.). The level of detail in the land use impact assessment needs to be defined in the goal and scope definition of the study.

Currently available data bases 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. If such a default impact assessment bases on world-average conditions, it would be attractive for users of fragile land areas to produce good LCA results by 'ignoring' the location of the actual land use. If a default impact assessment bases on worst-case conditions, there would be a strong motivation for LCA commissioners to 'know' where land use takes place.

**Box 1. Considerations on the impacts 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). This stresses 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.

# 6. 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 most relevant impact pathways identified in this paper affect the natural environment (biodiversity and some life support functions of soil not directly used by humans) and the natural resource aspect of soil quality (basically referred to biomass production) and biodiversity.

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 categories biodiversity and soil quality (midpoint approach) or to the damages on the natural environment and resources (endpoint approach). 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. The time lag between the land use intervention and 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.** We recommend using a *situation differentiation* (Jolliet 2005) 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; sealed soil –industry, residential, road-; 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 (EPD) 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 <u>for</u> the LCA practitioner, while a more specific (spatially differentiated) assessment will be done <u>by</u> the practitioner in 'land management LCA' studies.

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