III.3.3. LCI modelling consequences on the LCA results

The results presented in the last section depend occasionally on the models that have been used in order to determine the emissions. These models have uncertainties that are different from the uncertainty underlying any systems analysis (i.e.: the data uncertainty), because modelling uncertainties come into play. As modelling uncertainties can be controlled more directly with a review of the models applied, the main effects on the results are briefly commented below.

Modelling of the pesticide fractions

With respect to the way pesticide fractions entering different compartments are calculated some comments have to be made. Firstly, with respect to the pesticide emissions to air, it should be noted that the fraction going to air with wind drift is very small, compared to the fraction emitted to air through volatilisation from soil's and plant's surface. On the one hand, this fact confirms what is stated in the explanation of the wind drift modelling (section III.2.4), when it is suggested that the uncertainty underlying any value taken from Figure III-6 will be almost irrelevant from an environmental point of view. On the other hand, volatility appears as one of the key values determining the environmental fate of pesticides, and therefore any uncertainty within this parameter would significantly increase the uncertainty in the results. The uncertainty within volatility values is usually small, and wide ranges of volatility values are grouped in the α values (see Table III-20 and Table III-21). Nonetheless, the fact that these α values are in turn combined with predicted residence times increases the uncertainty.

As for the fraction of pesticide entering the water compartment, it must be noted that surface water is affected equally in all systems, with average surface runoff estimated from studies in field crops. This value might vary with site conditions (mainly slope and proximity to water courses), and therefore some degree of complexity should be added in the model in order to better predict f_w. The case of pesticide leaching to groundwater is the opposite: a model developed precisely for New Zealand conditions was used, and therefore the results can be regarded as highly qualified. Nevertheless, as it was pointed in the explanation of the results, the outcome of the model is strongly dependent on the soil type (as it was expected), and this could not be precisely defined for the Central Otago sites within PESTRISK. The reason is mainly the fact that the model does not content data for all the soil types in New Zealand. In this case, thus, inventory uncertainty on soil type is chiefly determining the results. Besides, the effects of weather (mainly rainfall) are included in the modelling through the database in PESTRISK. Here, the lack of specific data for Central Otago might have an influence on the results, although the effect of rainfall in irrigated systems is probably less important than the effect of soil.

Finally, the pesticide emissions to soil are usually very small, mainly due to biodegradation destroying most pesticide before soil leaves the system at harvest. Here two key points should be noted:

- First, the uncertainty in half-life values, which might be high due to different experimental conditions, will reduce the quality of the results. Indeed, many different values are often found for degradation half-lives of the same substance, and they present ample variability due to different conditions in the measurements. A variation of ca. ±20% has been considered for this study.

- Secondly, no effect of metabolites is considered in the modelling. Usually metabolites are less dangerous than the mother substance, but if this was not the case, an important weakness of the modelling might appear.

Emissions from fertilisers

In the case of fertiliser field emissions, the first obvious comment is for nitrate emissions, because they determine the effects on nutrification. A high uncertainty is to be expected in these emissions because no site-dependent modelling was done for their calculation. A nutrient balance perspective should be introduced in this issue, as it is expected that nitrate field emissions will anyway determine nutrification.

The nutrient balance should cover all other emissions related to nitrogen: N_2O, NO_x, NH_3 and CH_4. The emission estimates come from European literature, and relevant differences might be expected for some of them.

In the case of heavy metals emissions from fertilisers, some further refining of the model is needed, as they come from experiments on grain crops in Switzerland, France and the Netherlands. Particularly, uptake of heavy metals by fruit trees might be significantly different, and therefore have a crucial effect on the overall results. Nevertheless, it should also be noted that heavy metal field emissions from fertilisers do not seem to have a significant effect on the results, and mainly the emissions during production of several inputs (particularly machinery and pesticides) are relevant for the emissions of heavy metals.

III.3.4. Gravity and uncertainty analyses

A rough gravity analysis[21] has been performed during the description of the LCIA results (section III.3.2), by determining the LCI aspects that chiefly determine the results in each orchard for all the impact categories. This analysis is summarised in Table III-37, with the ranges of impact contribution caused by different impact sources. The confidence related to each one of the aspects mostly affecting the results is also done in Table III-37 under the column “Uncertainty analysis”.

[21] ISO 14042:2000 defines a gravity analysis as “a statistical procedure which identifies those data having the greatest contribution to the indicator result”.

Table III-37: Gravity and uncertainty analyses of the LCA results.

Impact categ.	**Site** [a]	***Gravity analysis***		***Uncertainty analysis***	
		LCI aspects mostly affecting the results	*% of contrib.*	*Aspects mostly reducing confidence (section where it is discussed)*	*Estimate of error range in LCI data (±%)*
Global Warming	IFP_HB_1	CO_2 emissions from energy consumption in field operations	48%	Figures for fuel consumption per hour (III.2.5, Table III-26 in page 89)	±10%
	IFP_HB_2		62%		
	IFP_HB_Avg		34%		
	IFP_CO_1		49%		
	IFP_CO_Avg		34%		
	OFP: all		73-80%		
	IFP_HB_1	N_2O field emissions from fertilisers	22%	Emissions might vary, but conditions are similar, and no big uncertainties are expected (III.2.3, Table III-15 in page 75)	Not relevant
	IFP_HB_2		7.5%		
	IFP_HB_Avg		47%		
	IFP_CO_1		30%		
	IFP_CO_Avg		48%		
	IFP: all	CO_2 emissions from pesticides production	5-11%	Total energy consumption in production (III.2.6)	±15%
	OFP: all	CO_2 emissions from machinery production	8-15%	Machinery's total use in lifetime (III.2.5, Table III-25 in page 88)	±40%
Human Toxicity through Air	IFP_HB_1	Benzene (and Pb) emissions from energy consumption in field operations	43%	Figures for fuel consumption per hour (III.2.5, Table III-26 in page 89)	±10%
	IFP_HB_2		35%		
	IFP_HB_Avg		49%		
	IFP_CO_1		48%	Benzene (as a hydrocarbon) emissions might be uncertain according to different references (III.2.8)	±25%
	IFP_CO_Avg		53%		
	OFP: all		80%		
	IFP_HB_1	Pesticide emissions by volatilisation	45%	α values and degradation half-lives introduce uncertainty to Equation 1 and Equation 2 (in page 83) [b]	±50%
	IFP_HB_2		58%		
	IFP_HB_Avg		35%		
	IFP_CO_1		36%		
	IFP_CO_Avg		28%		
Human Toxicity through Water	IFP: HB	Pesticide leaching	92-100%	HB soils are properly represented in PESTRISK	Not relevant
	IFP: CO			CO soils are not properly represented in PESTRISK [c]	±50%
	OFP: all	NMVOC and heavy metals emissions from energy consumption in field operations	66-80%	Figures for fuel consumption per hour (III.2.5, Table III-26 in page 89) NMVOC (as hydrocarbons) emissions might be uncertain according to different references (III.2.8)	±10% ±25%
	OFP: all	Heavy metals emissions from pesticides production	10-18%	Total energy consumption in production (III.2.6)	±15%
	OFP: all	Heavy metals emissions from machinery production	9-16%	Machinery's total use in lifetime (III.2.5, Table III-25 in page 88)	±40%

Table III-37: Gravity and uncertainty analyses of the LCA results. (continued)

Impact categ.	Site [a]	Gravity analysis: LCI aspects mostly affecting the results	Gravity analysis: % of contrib.	Uncertainty analysis: Aspects mostly reducing confidence (section where it is discussed)	Uncertainty analysis: Estimate of error range in LCI data (±%)
Human Toxicity through Soil	IFP: all	Pesticide residues in soil (mainly ziram, azinphos-methyl and metiram)	100%	Values of $\tau_{½}$ of microbial degradation present wide variability	±20%
	OFP: all	NMVOC emissions from energy consumption in field operations	60-80%	Figures for fuel consumption per hour (III.2.5, Table III-26 in page 89) NMVOC (as hydrocarbons) emissions might be uncertain according to different references (III.2.8)	±10% ±25%
	OFP: all	Heavy metals emissions from pesticides production	7-18%	Total energy consumption in production (III.2.6)	±15%
	OFP_HB_1	Copper emissions to soil from pesticide use	12%	Fraction of heavy metals remaining in soil (III.2.3, Table III-16 and Table III-17 in page 77)	±50%
	OFP_HB_Avg		21%		
Ecological Toxicity through Water (Chronic)	IFP_HB_1	Heavy metals emissions from energy consumption in field operations	14%	Figures for fuel consumption per hour (III.2.5, Table III-26 in page 89)	±10%
	IFP_HB_2		21%		
	IFP_HB_Avg		28%		
	IFP_CO_1		54%		
	IFP_CO_Avg		33%		
	OFP: all		42-52%		
	IFP: all	Heavy metals emissions from machinery production	15-34%	Machinery's total use in lifetime (III.2.5, Table III-25 in page 88)	±40%
	OFP: all		40-50%		
	IFP_HB_1	Heavy metals emissions from pesticides production	48%	Total energy consumption in production (III.2.6)	±15%
	IFP_HB_2		43%		
	IFP_HB_Avg		28%		
	IFP_CO_1		21%		
	IFP_CO_Avg		24%		
	OFP: all		7-15%		
Ecological Toxicity through Water (Acute)	IFP_HB_1	Heavy metals and cyanide emissions from energy consumption in field operations	17%	Figures for fuel consumption per hour (III.2.5, Table III-26 in page 89)	±10%
	IFP_HB_2		19%		
	IFP_HB_Avg		31%		
	IFP_CO_1		57%		
	IFP_CO_Avg		37%		
	OFP: all		46-56%		
	IFP: all	Heavy metals emissions from machinery production	14-30%	Machinery's total use in lifetime (III.2.5, Table III-25 in page 88)	±40%
	OFP: all		37-50%		
	IFP: all	Heavy metals emissions from pesticides production	16-25%	Total energy consumption in production (III.2.6)	±15%
	OFP: all		5-13%		
	IFP_HB_2	Azinphos-methyl and copper emissions from pesticide use	25%	Fraction leaving with run-off (III.2.4, section "Run-off")	±15%
Ecological Toxicity through Soil (Chronic)	IFP: all	Cyanide (and benzene) emissions from energy consumption in field operations	75-86%	Figures for fuel consumption per hour (III.2.5, Table III-26 in page 89)	±10%
	OFP: all		>90%		
	IFP: all	Pesticide residues in soil	<13%	Values of $\tau_{½}$ of microbial degradation present wide variability	±20%

Table III-37: Gravity and uncertainty analyses of the LCA results. (continued)

Impact categ.	*Site [a]*	*Gravity analysis*		*Uncertainty analysis*	
		LCI aspects mostly affecting the results	*% of contrib.*	*Aspects mostly reducing confidence (section where it is discussed)*	*Estimate of error range in LCI data (±%)*
Photochemical Oxidants Formation	IFP: all	NMVOC and CO emissions from energy consumption in field operations	90%	Figures for fuel consumption per hour (III.2.5, Table III-26 in page 89)	±10%
	OFP: all		>95%	NMVOC (as hydrocarbons) and CO emissions might be uncertain according to different references (III.2.8)	±25%
Acidification	IFP_HB_1	NO_x and SO_x emissions from energy consumption in field operations	54%	Figures for fuel consumption per hour (III.2.5, Table III-26 in page 89)	±10%
	IFP_HB_2		74%		
	IFP_HB_Avg		32%		
	IFP_CO_1		52%	NO_x and SO_x emissions might present slight uncertainties according to different references (III.2.8)	±10%
	IFP_CO_Avg		31%		
	OFP: all		73%		
	IFP_HB_1	NH_3 field emissions from fertilisers	12%	Emissions might vary, but conditions are similar, and no big uncertainties are expected (III.2.3, Table III-15 in page 75)	Not relevant
	IFP_HB_2		3%		
	IFP_HB_Avg		48%		
	IFP_CO_1		18%		
	IFP_CO_Avg		50%		
	IFP: all	NO_x and SO_x emissions from machinery production	6-11%	Machinery's total use in lifetime (III.2.5, Table III-25 in page 88)	±40%
	OFP: all		15-17%		
	IFP: all	NO_x and SO_x emissions from pesticides production	15-17%	Total energy consumption in production (III.2.6)	±15%
	OFP: all		9-11%		
Nutrification	IFP: all	NO_3^- field emissions from fertilisers	80-90%	Important variations in nitrate leaching are expected if a nutrient balance was performed in different locations (III.2.3)	±60%
	OFP: all		83-88%		
	IFP: all	NO_x emissions from energy consumption in field operations	10-17%	Figures for fuel consumption per hour (III.2.5, Table III-26 in page 89)	±10%
	OFP: all		11-16%	NO_x and SO_x emissions might present slight uncertainties according to different references (III.2.8)	±10%
Energy consumption	IFP: all	Energy consumption in field operations	68-76%	Figures for fuel consumption per hour (III.2.5, Table III-26 in page 89)	±10%
	OFP: all		83-90%		
	IFP: all	Energy consumption in machinery production	7-13%	Machinery's total use in lifetime (III.2.5, Table III-25 in page 88)	±40%
	OFP: all		8-15%		
	IFP: all	Energy consumption in pesticides production	9-13%	Total energy consumption in production (III.2.6)	±15%

[a]: sites within the same technology type (IFP or OFP) are only considered separately if contributions by the same substance in different sites show differences of at least 10%.
[b]: the relative error factor of applying Equation 1 and Equation 2 is expressed, rather than uncertainty factors for all the parameters in these equations.
[c]: the uncertainty margin has been calculated estimating leaching potential for the most problematic active ingredients in Central Otago (carbaryl, cyprodinil, triadimefon and tebufenozide) in 6 different soil types. The other parameters (weather, irrigation pattern, and dosage) have been left as in the analysis for IFP_CO_1 and IFP_CO_Avg (see Table III-22 in page 84).

Even though the application of uncertainty analysis[22] techniques to LCA is not properly formalised yet, a tentative estimation of error margins is done in order to define the degree of uncertainty in the LCA results. Combining both columns of the gravity and uncertainty analyses in Table III-37, Table III-38 is derived, including estimations of the error factors that should be considered for each impact category in each location.

The error propagation has been estimated according to how the variables containing uncertainty are introduced in the formulas for the calculation of the impact category contribution. The general formula to express the propagation of uncertainty may be found in any applied mathematics manual (see e.g. Heijungs 1996 for the application of uncertainty analysis in LCA). For a function with different variables $y=f(x_1,x_2,\ldots)$, the propagation of the absolute errors in these variables (Δx_1, Δx_2, …) to the error in the result of the function (Δy) is given by:

Equation 4: Propagation of absolute errors.

$$\Delta y = \left|\frac{\partial f}{\partial x_1}\right| \Delta x_1 + \left|\frac{\partial f}{\partial x_2}\right| \Delta x_2 + \ldots$$

It must be noted that the uncertainties given in Table III-37 are relative, and not absolute. Relative errors (δx_1, δx_2, …) can be expressed as:

$$\delta x = \frac{\Delta x}{\left|\bar{x}\right|}$$

The consequences of these equations are further explained in Table III-38's footnotes.

[22] The main idea behind uncertainty analysis is to determine if indicator results for the same impact category are significantly different from each other (ISO 14042:2000).

Table III-38: Error factors (±%) that should be considered for each impact category in each location of the study.

Impact category	*IFP*					*OFP*			
	HB_1	*HB_2*	*CO_1*	*HB_Avg*	*CO_Avg*	*HB_1*	*CO_1*	*HB_Avg*	*CO_Avg*
Global Warming	±6%	±8%	±7%	±5%	±5%	±14%	±14%	±14%	±14%
Human Toxicity Air	±43 [a]	±41%[a]	±35%[a]	±35% [a]	±33% [a]	±28%[a]	±28%[a]	±28%[a]	±28%[a]
Human Toxicity Water	n.r.	n.r.	±50%	n.r.	±50%	±41%[a]	±41%[a]	±41%[a]	±41%[a]
Human Toxicity through Soil	±52%[b]	±52%[b]	±52%[b]	±52% [b]	±52% [b]	±37%[a]	±31%[a]	±41%[a]	±31%[a]
Eco-Toxicity Water (Chronic)	±22%	±22%	±22%	±21%	±21%	±27%	±27%	±27%	±27%
Eco-Toxicity Water (Acute)	±17%	±21%	±21%	±19%	±19%	±28%	±28%	±28%	±28%
Eco-Toxicity Soil (Chronic)	±15%[b]	±15%[b]	±15%[b]	±15% [b]	±15% [b]	±9%	±9%	±9%	±9%
Photoch. Oxidants Formation	±32%[a]	±32%[a]	±32%[a]	±32%[a]	±32%[a]	±33%[a]	±33%[a]	±33%[a]	±33%[a]
Acidification	±18%[a]	±22%[a]	±17%[a]	±13%[a]	±13%[a]	±23%[a]	±23%[a]	±23%[a]	±23%[a]
Nutrification	±56%[a]	±56%[a]	±56%[a]	±56%[a]	±56%[a]	±54%[a]	±54%[a]	±54%[a]	±54%[a]
Energy consumption	±15%	±15%	±15%	±15%	±15%	±15%	±15%	±15%	±15%

[a]: when two values having relative uncertainty ranges (±x%) are multiplied, the uncertainties are added to express the uncertainty of the product.
[b]: the uncertainty of the fraction of pesticide remaining in soil after harvest is calculated from the product of the derivative of the degradation function (Equation 3, page 85) times the error in the variable "degradation half-life" ($\tau_{1/2}$) (see the text above and Equation 4). When expressed in relative terms, this error is proportional to $t_{harvest}$ and inversely proportional to $\tau_{1/2}$. On average, the factor multiplying the error in the biodegradation half-life is 2.59 for the pesticides chiefly affecting toxicity through soil: ziram, tebufenozide, cyprodinil, bupirimate, and triflumuron. Thus, the uncertainty of ±20% in $\tau_{1/2}$ (Table III-37) is amplified to ±52% through the propagation of the error in the formula for degradation.
n.r.: not relevant.

The uncertainty margins shown in Table III-38 may seem extraordinarily high for the non-LCA practitioner. Indeed, they tend to reduce the reliability of the LCA results for some impact categories. Nevertheless, they can be considered as normal in systems depending on so many inputs such as agriculture. Besides, a precautionary principle has been followed in the setting of uncertainty margins when no statistical information was available for the data (e.g.: for pesticide volatilisation an biodegradation half-lives). Indeed, one of the problems found for the completion of the uncertainty analysis is the lack of meta-data (i.e.: information about the data, or data quality information). Weidema & Wesnæs (1996) find similar uncertainty values in an example of application of data quality indicators to energy consumption figures in an agricultural LCA.

III.4. Interpretation of the apple LCA

In the light of the objectives set in the beginning of the apple LCA, the results of the LCIA and LCI phases may now be interpreted. First, the primary goal of the study (i.e.: detecting the environmental hotspots in integrated and organic apple production in two regions of New Zealand) is answered in section III.4.1. A discussion on the relevance of the different aspects affecting the results follows, which suggests that site-dependent parameters override the effect of region and technology on most LCA results. Besides, the contribution of several inputs production (machinery, pesticides, fertilisers…) to the results is given in this section, with some review of other references on the subject. After this first section, the secondary objectives described in III.1.2 are addressed: section III.4.2 presents some opportunities for improvement of the environmental impacts related to apple production in New Zealand, and section III.4.3 suggests research needs for a more generalised application of LCA in New Zealand's agricultural sector.
The analysis of the environmental hotspots in apple production points at a deep site-dependency for the results of this agricultural LCA; as the discussion of site-dependency was established as another primary goal of the study, this issue has been deeply developed in section III.5.

III.4.1. Environmental hotspots in apple production

A very deep and extensive knowledge has been gained on the apple production systems. Mainly a detailed picture of the aspects that generate the main impacts, and the processes through which they operate, has been obtained. Therefore, the primary goal of the study, i.e.: to detect the environmental hotspots dominating the different impact categories studied in the LCA of organic and integrated apple production, has been achieved. It must be noted that as normalisation and valuation scores were not available for New Zealand, no prioritisation of one impact category over the other can be done.

Contribution analysis
Table III-39 shows the issues from the life cycle inventory of apples that have the greatest influence on the results. These aspects are given for IFP and OFP systems, and they are illustrated with the relevance for every specific impact (with the range of relative contributions by the different producers in the study) and the main substances causing the impact through that aspect. It must be noted that hotspots are given both for IFP and OFP systems, even though the contribution to the impacts may be much higher in one of the production types. For instance, in human toxicity water and human toxicity soil OFP contributions are negligible compared to IFP, whereas in ecological toxicity soil and photochemical oxidants formation the impacts are higher in OFP. The significance of such differences between production types (organic or integrated) is shown in Table III-40, Table III-41 and Table III-42, and further discussed in section III.5. Discussion: Site-Dependency in Agricultural LCA.

Table III-39: Environmental hotspots for different impact categories in IFP (left) and OFP (right). Darker grey means higher relative contribution to impact category (see legend); main contributing substances are shown beneath the range of relative contributions by the different producers.

Category	Main contributors in IFP systems					Main contributors in OFP systems				
	Energy field emissions	*Pesticide field emissions*	*Fertiliser field emissions*	*Machinery production*	*Agro-chemicals [a] production*	*Energy field emissions*	*Pesticide field emissions*	*Fertiliser field emissions*	*Machinery production*	*Agro-chemicals [a] production*
Global Warming	**40-65% CO_2**		8-50% N_2O	5-12% CO_2	6-13% CO_2	**80-90% CO2**			8-15% CO_2	
Human Toxicity Air	**40-65% benzene**	**30-60% many a.i.**				**>95% benzene**				
Human Toxicity Water		**92-100% many a.i.**				**66-80% NMVOC, heavy metals**			9-16% heavy metals	10-18% heavy metals
Human Toxicity Soil		**100% many a.i.**				**60-80% NMVOC**	0-21% Copper			7-18% heavy metals
Ecological Toxicity Water (Chronic effects)	**14-54% heavy metals**			15-34% heavy metals	21-48% heavy metals	**42-52% heavy metals**			40-50% heavy metals	7-15% heavy metals
Ecological Toxicity Water (Acute effects)	**17-54% heavy metals & cyanide**	0-24% many a.i.		14-30% heavy metals	16-25% heavy metals	**46-56% heavy metals**			37-50% heavy metals	5-13% heavy metals
Ecological Toxicity Soil (Chronic effects)	**75-86% cyanide & benzene**	5-13% many a.i.				**>90% cyanide & benzene**				
Photochemical Oxidants Formation	**90% NMVOC & CO**					**>95% NMVOC & CO**				
Acidification	**30-75% NO_x & SO_x**		4-50% NH_3	6-11% NO_x & SO_x	6-15% NO_x & SO_x	**73% NO_x & SO_x**			15-17% NO_x & SO_x	9-11% NO_x & SO_x
Nutrification	10-17% NO_x		**80-90% NO_3^-**			11-16% NO_x		**83-88% NO_3^-**		

Legend for Table III-39										
Highest contribution from a producer to the impact category:		>75%		51-75%		26-50%		11-25%		≤ 10%

[a]: includes insecticides, fungicides, herbicides, thinners, and calcium products production.

From Table III-39 a general idea on where the impacts come from in IFP and OFP can be gained. The first conclusion arising from the table is that most impacts are directly dominated by producer's practices in some way or other: election of fertiliser or pesticide active ingredients, efficiency in the use of machinery, etc. Site conditions (particularly soil type) have a significant effect on the impact categories as well, mainly through their effect on field emissions. Besides, it can be stated that integrated production presents a wider variety of impact sources than OFP, and these are both related to energy emissions and field emissions from pesticides and fertilisers. In the case of organic apple production, a clear focus of impact generation can be seen in the energy consumption, which always shows the highest contributions except in the case of nutrification. This is because inputs used in organic fruit production are in principle less problematic than those used in IFP (less toxic substances are used for pest and disease management in OFP, and non-soluble fertilisers are used, thus reducing the possibility of loss from the system). Apart from this overall distribution of impact sources, ample variations appear in the relative contributions of each producer's item to the impact categories.

Aspects affecting the LCA results

The hypotheses underlying the goal definition were that different technologies (integrated or organic) do have an effect in the agricultural LCA results, and that further effects might arise from the region where agriculture is taking place. In order to check whether the variations found in the results are due to technology and/or region effects, and not on particular producer's practices, a two-way analysis of variance (anova) was done on the results. Analysis of variance is a technique for partitioning the variance in a set of data in such a way that contribution of these partitions to the overall data set can be assessed. In the case of the apple LCA, a factorial design was chosen for the anova, in order to check whether the factors "technology" and "region" had a bigger effect on the results than the mere variation between the different sites under study. That is, whether variances between technologies or between regions (treatment variances) were bigger than variance within those groupings (error variance).

Table III-40: Results for the analysis of variance of the apple LCA results.

Impact category	Significance for (p) [#]		
	Technology	Region	Technology · Region
Global Warming [a]	n.s.	n.s.	n.s.
Human Toxicity Air [b]	•	n.s.	n.s.
Human Toxicity Water [b]	**	n.s.	n.s.
Human Toxicity Soil [b]	***	n.s.	n.s.
Ecological Toxicity Water (Chronic effects) [a]	n.s.	n.s.	n.s.
Ecological Toxicity Water (Acute effects) [a]	n.s.	n.s.	n.s.
Ecological Toxicity Soil (Chronic effects) [b]	*	n.s.	n.s.
Photochemical Oxidants Formation [b]	**	n.s.	n.s.
Acidification [b]	n.s.	n.s.	n.s.
Nutrification [b]	n.s.	n.s.	n.s.
Energy consumption [b]	*	n.s.	n.s.

[#]: n.s. (not significant) = p>0.1 ; • (marginally significant) = 0.05 < p ≤ 0.1; * = 0.01 < p ≤ 0.05; ** = 0.0001 < p ≤ 0.001; *** = p ≤ 0.0001.
[a]: impact categories following a normal distribution.
[b]: impact categories following a lognormal distribution.

From Table III-40 it is apparent that no regional effects could be detected in the apple LCA. It must be taken into account that this result may be due to the small number of replicates, although it was expected that no big differences would arise from regional characteristics, as it has been explained throughout the text. On the other hand, technology choice seems to heavily influence the contributions to some impact categories, namely: human toxicity (both through water and soil, and marginally through air), eco-toxicity through soil, photochemical oxidants formation, and energy consumption.

When results of the anova tell us that the variance due to technology and/or region (treatment variance) is not bigger than the error variance, it actually means that the results depend more on some parameter specific for the sites under analysis. In other words, it can be said that those impact categories are site-dependent. Nevertheless, this will only give information on the impact categories that have been found not to depend on the factors used in the anova (technology and region).

Accordingly, results are considered site-dependent when differences between contributions within the same technology group to an impact category are above the error margins considered for that category (see Table III-38, in page 124). Table III-41 is constructed in this way: each row shows the results for the category indicator followed by letters. Values for different sites in the same impact category followed by the same letter have overlapping error margins for that impact category.

The results of such analysis of the main trends for site- and technology-dependency in the results of the apple LCA are shown in Table III-42. In this table, also the main reasons for site-dependency are included as brief notes in parenthesis. When an impact category is found to be technology-dependent in the anova (Table III-40), the technology type having the worst contribution to it is also shown in parenthesis.

Table III-41: Impact assessment results for each impact category and site (f.u. = 1 ton of export and local quality apples). Figures followed by the same letter have differences within the error margins for that impact category.

	IFP					OFP			
	HB_1	*HB_2*	*CO_1*	*HB_Avg*	*CO_Avg*	*HB_1*	*CO_1*	*HB_Avg*	*CO_Avg*
Global Warming	43.45	35.24	66.51	95.58	94.60	104.44	70.21	66.31	65.04
kg CO_2/f.u.	b	a	c	d	d	d	c	c	c
Human Tox Air ($\cdot 10^6$)	19.21	21.95	16.29	23.81	20.61	45.68	23.35	24.83	24.02
m^3 air/f.u.	a	a	a	a	a	b	a	a	a
Human Tox Water	6787.04	5105.86	35050.85	1679.97	4604.80	303.59	210.83	199.40	195.63
m^3 water/f.u.	d	c	e	b	cd	a	a	a	a
Human Tox Soil	207.46	1609.75	1828.29	274.13	505.66	0.79	0.43	0.54	0.42
m^3 soil/f.u.	b	c	c	b	b	a	a	a	a
ExoTox Wat Chron	851.62	770.76	1315.66	1196.06	1182.78	1345.73	1295.10	1092.23	1037.57
m^3 water/f.u.	ab	a	b	b	ab	b	b	ab	ab
EcoTox Wat Acu	82.61	95.45	130.88	117.38	115.41	134.09	128.67	108.19	102.91
m^3 water/f.u.	a	ab	b	ab	ab	ab	ab	ab	ab
EcoTox Soil Chron	16.59	16.91	25.78	25.89	24.82	54.57	35.93	34.37	33.73
m^3 soil/f.u.	a	a	b	b	b	d	c	c	c
Phot. Oxid. Form.	0.04	0.04	0.04	0.06	0.06	0.18	0.09	0.09	0.09
kg C_2H_4/f.u.	a	a	a	ab	ab	c	b	bc	bc
Acidification	0.32	0.35	0.41	0.79	0.76	0.48	0.56	0.41	0.38
kg SO_2/f.u.	a	a	ab	c	c	ab	b	ab	ab
Energy Consumption	453.58	448.37	570.27	646.14	620.15	1379.03	887.88	852.53	830.75
MJ/f.u.	a	a	ab	bc	b	e	d	cd	cd

Table III-42: Dependency of LCA results on site and technology characteristics.

Category	*Dependency*	
	Site (causes)	*Technology (worst)*
Global Warming	YES (machinery intensity; type of fertiliser used)	NO [a]
Human Toxicity Air	YES for OFP [b] (machinery use intensity)	NO
Human Toxicity Water	YES for IFP (election of a.i.; soil type; irrigation; timing…)	YES (IFP)
Human Toxicity Soil	YES for IFP (election of a.i.; timing of application)	YES (IFP)
Ecological Toxicity Water (Chronic effects)	YES for IFP (machinery use intensity; irrigation patterns; type of fuel)	NO
Ecological Toxicity Water (Acute effects)	YES for IFP (machinery use intensity; irrigation patterns; type of fuel; election of a.i.)	NO
Ecological Toxicity Soil (Chronic effects)	YES (machinery use intensity)	YES (OFP)
Photochemical Oxidants Formation	YES for OFP [b] (machinery use intensity)	YES (OFP)
Acidification	YES for IFP (machinery intensity; type of fertiliser used)	NO
Nutrification	*No conclusions should be drawn for this impact category (see text)*	
Energy consumption	YES (machinery intensity)	YES (OFP)

[a]: but the source of impacts does depend on technology (see Table III-39).
[b]: but only due to Heather Gregory's orchard (OFP_HB_1) because of the extraordinary energy consumption.

No conclusions should be drawn for the impact category Nutrification, because as it was pointed out in the results (see section III.3.2 under Nutrification) it totally depends on a reference value for nitrate leaching. Nevertheless, as nitrate emissions to groundwater will usually be of the same order of magnitude than the figure considered in this study (33.1 kg $N \cdot ha^{-1} \cdot year^{-1}$), it may be noted that the overall contribution to this impact category will be probably always determined by a single process: nitrate leaching. Therefore, this process should be modelled using a nutrient balance perspective for each orchard. When an orchard performs a sound nutrient management, aimed at reducing nutrient losses through leaching, energy-related emissions (mainly NO_x) will come into play (and the overall contribution of that orchard to nutrification will then be lower than other orchards). As nitrate leaching depends on many parameters related to the site (soil type, weather, and irrigation practices), this impact category will surely be site-dependent. Also the amount and type of fertilisers will be determined by farming practices, which will increase site-dependency. The technology type (IFP or OFP) will also determine nutrient management, and so nutrification might also be technology-dependent. However, this conclusion is not as straightforward as the former, because the nutrient balance is dominated by the “natural” vegetation turnover (grass clippings, falling leaves and prunings, see section III.2.3).

As it can be seen in Table III-41 and Table III-42, most impact categories are site-dependent to some degree (i.e.: different letters are found within the same technology group for these impact categories). In organic production systems, some impact categories are not site-dependent[23] and some others[24] present huge differences in contribution only when compared to a single producer: Heather Gregory (OFP_HB_1), who consumes much more energy than the others. Site-dependency is more frequent in IFP sites. As the discussion on site-dependency was among the primary goals of the study, it is given further attention in section III.5.

Technology-dependency: Environmental preference of organic or integrated fruit production

In the case of technology-dependency, four impact categories have been found to depend significantly on the technology type according to analysis of variance: Human Toxicity through Water and Soil (more affected by IFP orchards), and Eco-toxicity through Soil and Photochemical Oxidants Formation (dominated by OFP orchards). The indicator on energy consumption is also dominated by organic orchards.

Even though the comparison of organic and integrated apple production was not a primary goal of the study, in any environmental analysis including organic and integrated (or intensive/ conventional) sites it is unavoidable to address it at one point or another. First, nevertheless, it must be said that no concluding answer may be given in the light of the results presented in last section. Mainly, this is because no normalisation or valuation was performed (because the aims of the study did not need it), and therefore no preference can be put on one impact category or another. As IFP and OFP generate different environmental impacts (i.e.: some impact categories

[23] Human toxicity through water; human toxicity through soil; eco-toxicity through water (both chronic and acute effects); acidification.

[24] Global warming; human toxicity through air; eco-toxicity through soil; photochemical oxidants formation; energy consumption.

are dominated by IFP sites and some other are dominated by OFP sites, with many impact categories not showing clear differences between both systems), no general conclusion can be drawn.
This is at the same time a very important answer: there is no clear environmental preference for organic apple production as it could be expected. The high (non renewable) energy consumption in organic sites is the main reason for this. On the other hand, organic orchards are clearly preferable on some toxicity impact categories due to the avoidance of synthetic pesticides, but this only holds true for those toxicity impact categories where pesticides play a key role: human toxicity through water and soil. The elevated energy consumption and the use of some long-lived fungicides (namely copper substances) give surprisingly high contributions to OFP in some toxicity impact categories (particularly ecological toxicity through soil).

Region-dependent aspects in the LCA results
Finally, no significant dependency on the region has been found in the LCA results. Nevertheless, some key differences arising from region-dependent aspects have also been detected; these are mainly related to differences in pesticide use, and do not represent significant changes in the overall LCI results. As a site-dependent LCIA was not used, the effects of region on the results have not been assessed. Indeed, regional aspects might have been very important if some impact categories had been assessed: chiefly water consumption leading to depletion and impacts on biodiversity (see section III.5.2).

Environmental relevance of inputs production for apple growing
Audsley *et al*. (1997) suggest that machinery production plays a relevant role in agriculture, and report a share in total energy consumption of 13-37% in arable systems of different degree of mechanisation. The present study has confirmed this, and the results show that the contribution of machinery production to the overall impact may be even higher in some specific impact categories, even though the share in the energy consumption indicator is somehow smaller: 7-15%. Thence, while machinery production is responsible for 5-15% of Global Warming and Acidification, the share rises up to 50% in impacts caused by emissions of heavy metals to water (e.g.: eco-toxicity through water, see Table III-39). The effect of machinery production on the LCA impacts tends to be higher in organic systems, due to the higher machinery intensity and in some occasions to the lower yield.
In the case of pesticides production, it is relevant in the same impact categories as machinery production, and with similar contributions (see Table III-39). In this case, integrated systems show a higher share from pesticide production in relation to OFP, due to the higher use of pesticides in IFP. Audsley *et al*. (1997) find that pesticide production is responsible of ca. 6% of total energy consumption in a high-input system, which represents a smaller impact than that found in the present study for the energy consumption indicator (11-18% in IFP) and energy-related impact categories (global warming, eco-toxicity and acidification). Probably, the higher share for pesticide production in apple production than in wheat production (presented in Audsley *et al*. 1997) is due to a higher pesticide input in fruit production, and to higher energy consumption in arable crops field operations (e.g.: ploughing operations require a lot of energy). On the other hand, Stadig (1997) states that pesticide production is not relevant at all in an apple LCA. Nevertheless, it must

be noted that he includes the transportation phase of apples (cradle-to-store instead of a cradle-to-gate approach, such as in this study and in the wheat case studies reported by Audsley *et al*. 1997), which might diminish the relevance of pesticide production.

Fertiliser production represents less than 3% of energy consumption in all systems, and the contribution to all impact categories is almost irrelevant. Therefore, fertilisers are only important from the point of view of field emissions, as it has been discussed in the last sections (see particularly Table III-39).

III.4.2. Improvement opportunities in apple production in New Zealand

According to the results found in the apple LCA, the overall objectives of improvement opportunities should be aimed at:

- reducing energy consumption;
- reducing toxicity of the active ingredients used or reduce their emissions;
- reducing nutrient losses to air and groundwater;
- improving machinery use efficiency.

With these goals in mind, some concrete options for reducing the environmental impacts may be derived from the LCA results and a little bit of imagination (Table III-43). These are described in the next paragraphs, and whenever it is possible not only a description of the option and its environmental advantages is given, but also the possible trade-offs and drawbacks for the farmer.

Table III-43: Improvement objectives accomplished by different improvement opportunities in apple production.

Improvement opportunities	***Improvement objectives***			
	Reduced energy consumption and impacts	***Reduced toxicity and emissions of agro-chemicals***	***Reduced nutrient losses***	***Increase machinery use efficiency***
Sheep grazing of the understorey	✓	✓	**?**	✓
Use of blossom burners for fruit thinning		✓		
Machinery rental				✓
Use of biodiesel	✓**?**			
Farm management for reduced nutrient loss	**?**		✓	
Mapping areas sensitive to pesticide leaching		✓		
Basing irrigation on a water balance	✓	**?**	**?**	
Indicator set for environmentally sound farm management	✓	✓	✓	✓
Information, training, and advice to farmers	✓	✓	✓	✓

Sheep grazing of the understorey

If sheep flocks were allowed to graze between tree rows they could feed on the understorey's grass and weed, and as long as sheep did not eat the fruit nor damage the trees, it would be an environmentally sound option for understorey management. This would reduce the economic and environmental cost of mowing and fertilising (thanks to return of sheep manure to soil), and would avoid the need for herbicide use. Indeed, sheep may provide, with appropriate stocking densities, a cheap and efficient way of mowing and controlling weeds, as well as providing a source of nutrients and organic matter through dung (Beaufoy 2001). Besides, this would provide an extra output from the apple orchard (forage); some burdens of the land occupation and the part of irrigation directly needed for grass should be allocated to this extra function, thus further reducing environmental impacts of apple production. It must be noted that ammonia emissions coming from dung should be allocated mainly to sheep growing but possibly also to apple production. This option would also increase soil quality in the sense of increasing soil organic matter (and thus increase fertility, reduce erosion risk, improve soil structure, etc.), although this has not been explored yet in LCA (see Chapter IV).

No sheep were observed to be grazing in any of the systems analysed, even though the closeness of sheep-raising areas to apple orchards might facilitate this practice. Possibly the need to coordinate apple producer's and sheep farmer's activities would be one of the key difficulties for this option, and therefore a lot of effort should be put on communication.

Use of blossom burners for fruit thinning

Fruit thinning options are currently the matter of research in OFP orchards, where spraying salt and lime sulphur solutions to apple trees during flowering reduces the need of hand thinning. As in New Zealand hand thinning is performed with the help of hydra-ladders, this reduction would significantly help in reducing fuel consumption (thinning operations are responsible of around one third of hydra-ladder use in OFP orchards, see Table III-14 in page 68). This option would not probably mean any improvement in a system not as mechanised as the systems analysed in this study. Thence, for example, in a poor country with no intensive mechanisation the transition to blossom burner-based thinning will mean a reduction on expenditure on labour, but not on environmental impacts. LCA could be used to monitor and assess the effects of such a change.

In IFP systems, the transition to less toxic agro-chemicals would mean reduced contributions to some impact categories (particularly Human Toxicity through water where a significant contribution from carbaryl is observed, see Figure III-12 a). Therefore, if alternative fruit thinners prove to be as effective as carbaryl or naphthylacetic acid, farmers should consider changing to this new option.

On the other hand, salt and other emissions to soil should be modelled in order to prevent negative effects such as salinisation. Apart from this issue, no negative effects on the farmer are to be expected, because this option fully relies on the machinery that is already available in every orchard.

Machinery rental

Production of machinery has been demonstrated as a relevant source of impacts despite the elevated uncertainty underlying values for total use in lifetime. Therefore, action should be taken on both reducing machinery use intensity and reducing the impacts allocated to each machine. The latter option deals on the one hand with the construction and design of the machines (e.g.: use less material, or recycled materials…), and on the other hand with the machinery use efficiency. As machinery design is out of the scope of this study (and out of the reach of farmers), no comments will be made on that issue. However, in the case of use efficiency there is some scope for improvements to be made. If each machine was used more intensively, the impacts per hour of use would be reduced; i.e.: the total use in lifetime of a machine must increase. As most machines are only used occasionally, there is a possibility for renting the machines; with rental, the total use in the lifetime of the machine increases and thus impact per hour of use decreases, as well as the number of machines used in the field in a broader sense (e.g.: in New Zealand).

Indeed, this practice was observed in one orchard: Mr. Rue Collin –IFP_HB_2- used to rent a special motorbike for spraying herbicides. This fact could not be included in the LCI due to the lack of figures for total use in lifetime of the motorbike. Even though this option might seem unreal in the case of tractors or hydra-ladders (because they are already used intensively, and for some operations such as harvesting everybody would need them at the same time), it is a sound option for those machines used seldom: mower, mulch-mower, fertiliser spreaders, weed-eater, etc. Especially the mulch-mower seems to have a significant influence on machinery-derived impacts in the case of OFP (see discussions in section III.3.2, especially for global warming and eco-toxicity through water).

Renting machines usually reduces costs for the farmer, as maintenance costs and depreciation are shared amongst more farmers. Nevertheless, it requires some more planning and the existence of renting companies that are flexible enough with dates (to allow for raining days, when some operations cannot be performed) and can cope with the farmers' needs.

Of course, what is intended by this option is that the total use in lifetime of the machine increases (see Table III-25 in page 88). The lifetime in years of the rented machine might decrease due to its more intensive wear, and thus it should be monitored that the higher use per year compensated this shorter lifetime. Besides, data quality for total use in lifetime should be improved in order to be able to make sound decisions. A closer work with machinery retailers and farmers is needed in this issue.

Use of biofuel

Emissions from fuel combustion are the main responsibles of many of the environmental impacts that have been analysed (global warming, human toxicity through air, eco-toxicity through water and soil, photochemical oxidants formation, acidification). A change in fuel type may thus lead to reduce contributions in all these impact categories. Particularly, biodiesel will probably have reduced emissions of heavy metals and SO_x, thus improving the system's contribution to eco-toxicity through water and soil and acidification. In addition, the CO_2 emitted by biodiesel combustion is "renewable" (from a short bio-geo-chemical cycle), and thus not considered as contributing to global warming. However some checks should be made in order to assure that reduced energy efficiency due to lower optimisation of engines for this type of fuel, or increased emissions do not counter-balance the environmental performance of biodiesel. Particularly, emissions of volatile organic compounds (VOC) and particularly benzene should be carefully monitored because of their environmental relevance.

Such an improvement opportunity, though, has profound effects on the socio-economical context of the farm. Chiefly, a proper provision of biodiesel should be assured if farmers are to rely on it, unless engines may switch between fuel types without any problem. This would require that New Zealand or neighbour countries destine enough land extension for the production of such fuels. Apart from the provision of biofuel, a proper distribution is also required, and thus petrol stations should be prepared to sell biofuel as well as normal diesel during a certain adaptation period. These considerations lie beyond the scope of the thesis, but should be taken into account before any of the improvement opportunities are put into practice.

Aiming farm management at nutrient loss reduction

This is a very complex issue indeed, because agronomic requirements and goals might collide in some occasions with environmental ones. Several options do appear with the overall objective of reducing nutrient emissions (both to air and groundwater), and therefore a sound and detailed analysis should be performed in order to determine which one has most potential for improvement. Some of the options that can be considered are:

- Promoting the growth of the understorey in order to absorb the excess of nitrogen. Nevertheless, as the understorey is also responsible of returning a big amount of nutrients to the soil (see section III.2.3, under Nitrate emissions), a method should be developed to avoid this return of nutrients to increase leaching. A possibility here would be not to directly apply herbage clippings to soil, but collect them and compost them off-site with a careful control of NH_3 emissions, in order to stabilise nitrogen. This could also be applied to prunings.
- Promote an environmentally sound fertiliser use, depending on the time of the year. In section III.2.3 it can be seen that field emissions from fertilisers depend on the timing of their applications, and that some fertilisers have bigger emissions during cold seasons while others have bigger emissions in high temperature. Thence, one type of fertiliser or another should be recommended depending on these issues (e.g.: to reduce N_2O emissions, urea and ammonium fertilisers should be used only in cold months, while nitrate fertilisers seem more appropriate for hot periods, see Table III-15). Also fertilising aimed at reduced ammonia emissions should be studied and promoted.

Creating a map of areas sensitive to pesticide leaching

Pesticide leaching may determine the impacts to human toxicity through the ingestion of groundwater. As the soil type chiefly determines pesticide leaching, it is possible to make a rough map of the most sensitive areas to pesticide leaching, with the aid of a soil map. This could be used for planning, e.g.: in order to determine areas where only organic farming should be performed. In order to elaborate such a map, a model such as PESTRISK may be used by completing its soil database and performing a standard check including the most problematic pesticides (e.g.: carbaryl, lufenuron, triflumuron, chlorpyrifos, cyprodinil, etc.). Targets for maximum pesticide leaching should then be set and then prevent the use of toxic substances in areas that would probably exceed these targets.

Basing irrigation on a water balance perspective

The huge differences observed between different producers indicate that orchards are watered without much agronomic foundation. If irrigation was done on a water balance perspective, important opportunities for water saving would appear. Possibly, also leaching of agro-chemicals and nitrate would be reduced because of the smaller amount of water percolating through the soil.

In addition, the energy consumption by irrigation has great variability, which gives an ample scope for improvements.

Developing an indicator set to guide an environmentally sound farm management

The LCA results should be developed into a set of reporting indicators that can be easily monitored and implemented by the farmers themselves, or technicians from the administration, and which are relevant for the environmental sustainability of fruit production. The idea is to create a reporting procedure so that the evolution of the most significant sources of impact can be followed, and actions be taken to reduce negative trends in time. Even though the development of these indicators deserves a special research program, examples of such indicators could be:

- use or absence of specific and highly toxic active ingredients (which is already monitored through the spray diaries);
- kg of active ingredients used per tonne or per hectare, weighted according to their environmental significance;
- type and amount of fertilisers being used (which would provide an indication of the likely air emissions);
- fuel consumption and/or indicators of (specific) machinery use, such as number of mowing events per year, hours of hydra-ladder per hectare, etc. (in order to control impacts arising from this source);
- etc.

These indicators should be used to guide farming activities, and should be a part of a specific information and training program (see below).

Information, training and advice to farmers

Farmers' practices have proven to be keystones in the environmental performance of different apple orchards. Farmers have indeed the capacity to change the environmental results of the LCA analysis more than changes in production techniques for agro-chemicals or even farming machinery. Therefore, almost all of the improvement opportunities that are cited in this section need the cooperation of farmers in order to succeed. In turn, a change in farmer's practice may lead to important improvements in all the objectives set for the improvement opportunities (see Table III-43).

New ways of communication should be developed to inform the farmers on the LCA results and the improvement opportunities. Indeed, it is difficult to communicate procedures that are usually against "what has always been done", and still get the positive reaction that is needed if the environmental performance of agricultural production is to improve. In this respect, connecting the LCA results and the socio-economical needs of companies will be crucial for the farmers' acceptance. For instance, if they see LCA as a way to increase their competitiveness, to facilitate legislation compliance or the communication with the rest of the supply chain, there are more opportunities to get a positive reaction. Therefore, besides of an improvement opportunity, this action is also a need of further research.

III.4.3. Needs of further research

From Table III-39, it is interesting to note that in OFP systems, most impacts are caused by energy consumption or inputs (machinery, pesticides…) production. On the other hand, IFP systems have many problems related to substances crossing the field's boundaries: emissions to air, groundwater and soil from fertilisers and pesticides. The modelling of such processes thus becomes a key issue for the credibility of LCA results in IFP, while mostly the data quality for inputs consumption (chiefly energy, but also machinery and pesticides) will determine the results of LCA of organic apple production. These facts also have a direct translation in the research needs for a more generalised application of LCA to NZ agriculture.

Multidisciplinary approach within New Zealand research institutes to allow for site-dependency

First, it must be noted that the economic importance of agriculture in New Zealand, and thus the research effort that is already put onto agricultural activities, has enormously facilitated the conduction of the present LCA. Indeed, the fact that spray diaries are collected for all producers in New Zealand is a key issue to allow for a high quality data collection in pesticide use. Besides, the existence of a network of research institutes on agriculture provides with a valuable knowledge source: e.g. PESTRISK (a model from HortResearch) has made it possible to accurately model the fraction of pesticide likely to leach from the system, using New Zealand conditions. Particularly the research areas within HortResearch represent a potential facilitation for a detailed application of LCA to New Zealand agriculture. Actually, one of the issues that need further research as detected by the LCA study, nitrate-leaching risk, is currently studied by HortResearch staff, and some applications of the SPASMO computer model for nitrate leaching have already been performed (see e.g.: Green *et al*. 1999; 2001). The possibilities of a multidisciplinary approach to LCA conduction should thus be further evaluated and explored, and these should be based on the efforts already made. For instance, as site conditions have proven to determine the LCA results, the databases for soil conditions and weather patterns should be improved for the application of models such as PESTRISK and SPASMO. Other disciplines should also come into play, such as mechanical engineers for a more refined assessment of machinery use in the farm and machinery production (see below).

Now that the application of LCA to New Zealand apple production has been further consolidated, new agricultural sectors should be addressed.

Reduction and control of uncertainty

Meta-information is required to facilitate data uncertainty analysis, as the uncertainties found are big (see Table III-38) and limit the extent of the conclusions. As uncertainty factors for many data were established in a rather conservative way, estimates of higher and lower expected values should be found for each one of the data commented in Table III-37 (mainly for the values of biodegradation half-lives, total use in lifetime of machinery, pesticide volatilisation (α values), and hydrocarbon and CO emissions from tractors). Particularly the machinery use presented big uncertainties, due to the lack of data on real machinery use. As this has proven to be very relevant for the LCA results, some effort should be put to monitor how machines are used in different farms. This would allow checking the environmental soundness of improvement opportunities such as machinery rental. Therefore, a specific section on machinery use should be included in the data collection sheet for future agricultural LCA studies. In addition, the temporal quality of some figures should be improved, such as data for energy consumption in machinery and pesticide production.

Besides, model uncertainty should be reduced in some cases. This is the case for nitrate leaching (where no model was actually applied, but a reference value was used), pesticide leaching, and heavy metals from fertilisers and pesticides remaining in soil. These models should be further explored and possibly refined and adapted to New Zealand conditions in order to reduce the uncertainty margins.

The reduction of uncertainty might have an influence on some of the LCA results. Indeed, some impact categories have not been found to be site-dependent in Table III-42 (page 129) because the error margins were bigger than the *a priori* big variances. If the error margins are reduced, the differences might become more significant and thus suggest a stronger effect of the site dependency.

Completion and furthering of the life cycle analysis

The boundaries of the life cycle analysis were set at the farm gate because the main goal of the study was to address the complications arising from agricultural LCA. Therefore, the study should be completed down to the grave, even though this has not been a general practice in agricultural LCA (Audsley *et al*. 1997; Cowell & Clift 1997; Cowell 1998). At least, it is highly relevant to continue the study until the consumer's door, as the overseas transportation may represent a highly significant contribution to the energy consumption in New Zealand agriculture. Stadig (1997) already mentions that transportation's energy consumption overrides all other sources of energy consumption in apples produced in New Zealand, and this should be further investigated in order to find opportunities for improvement.

On the other hand, the scope of the study should be furthered with more sites in order to properly assess site-, technology- and region-dependency. As this was not initially a goal of the study, only some sites were included. More sites from different regions and covering both IFP and OFP should be randomly selected and analysed following the same structure presented in this study. In this way, site-dependency would be more properly checked, now that this has proven to be a relevant source of variation.

Criteria for organic agriculture

Apart from the toxicity aspects and the care for soil quality, some attention should be put on the farmer's energy consumption if organic agriculture practices are to be environmentally sustainable. In rich countries, mechanisation is substituting for human labour, and this hampers the environmental sustainability of the farm. Of course, further research incorporating social and economic information needs to be performed before any decisions are made on this issue.

This discussion raises still another point, of semantic nature, which is the name organic agriculture should have. Indeed, a variety of names has traditionally been used for this type of agriculture: the European Council Regulation No 2092/91 of 24 June 1991 on organic production of agricultural products mentions the different expressions used in the European Union countries to refer to this type of agriculture and its products. In English language, these correspond to “ecological agriculture” (in Spanish, Danish, German, and Swedish), “biological agriculture” (in French, Greek, Italian, Dutch, and Portuguese), and “organic agriculture” (only in English). Even though it is not the aim of this dissertation to start a discussion on the appropriateness of such names, it should be clear from the results presented so far that certification bodies should include criteria on the energy expenditure in field operations in order to properly call it “ecological” agriculture. The terms “organic” and “biological” seem to be more consistent with the aims of such agriculture, but the scope should nevertheless include energy-related aspects. According to the New Zealand apple LCA results, the terms “low input agriculture” and “extensive agriculture” would even be less appropriate than “organic agriculture”. With all this, though, I do not want to make the point that OFP is worse than IFP from an environmental point of view, or even that OFP is not environmentally friendly at all. Indeed, these statements could not be made because the LCA lacks the steps of impact assessment needed to do this kind of assertion: the normalisation and valuation steps. The only comment that needs attention is that current criteria for organic agriculture do not cover so far important aspects that hamper the environmental sustainability of this type of agriculture.

Inclusion of soil quality, biodiversity and water depletion in LCIA

On the other hand, soil quality and biodiversity, which are actively addressed and protected by organic agriculture, have not been included in the analysis due to LCA methodology gaps. Therefore, no sound comparison could still be made of integrated and organic apple production, and these issues should be further developed in order to fairly compare both technologies. Besides, water depletion is another issue not commonly included in LCA. Even though data for water consumption was collected for all the systems under study, no further characterisation could be performed based on the origin of this water. As this is another important issue for the comparison of agricultural sites, mainly in arid regions, it would be interesting to address it in future applications of LCA. The aspects relating LCIA and site-dependency are further addressed in section III.5.2.

III.5. Discussion: Site-Dependency in Agricultural LCA

There is a continuing debate within the LCA community on whether site-dependent data should be included in LCI, and whether LCIA should be site-dependent (Cowell 1998, p. 50). Actually, the degree of site-dependency in an environmental analysis depends on the type of decision that is being made: whether it represents a choice of site (and then Environmental Impact Analysis is a typical tool), or a choice of technology (and then LCA is usually mentioned as a convenient tool). The election and performance of different technologies in agriculture are also affected by the site, and therefore the inclusion of site-dependent data seems obvious. Also the SETAC Working Group on LCIA (Udo de Haes *et al*. 2002) mentions the issue of site dependency for the impact assessment phase of LCA, when discussing about the Generic Application Dependency that should be allowed in impact assessment methodology.

In the LCA case study of apple production in New Zealand, results have been found to be highly site-dependent. Particularly the effects of site-dependency on the LCI results were detected, and these are discussed in section III.5.1. Some comments can also be done on the LCIA aspects, which were not covered in the apple LCA case study; these are provided in section III.5.2.

III.5.1. Site-dependency in the Inventory Analysis results

From Table III-39 and Table III-42 the main inventory aspects that affect the LCA results can be drawn, and broadly classified into the following three categories:

- technology type (integrated vs. organic)
- technique (producer's practices)
- physical conditions of the site (soil type and climate / weather)

Of these aspects, technology is what has traditionally been regarded as the main object of comparison (and thus the main source of difference) in LCA. As it has been concluded in section III.4.1, the general definition of agricultural technologies (e.g.: integrated or organic) is not enough to predict most environmental impacts arising from agricultural production. Indeed, description of particular producer's techniques within a general technology is necessary for an overall prediction of the environmental impacts. These practices are furthermore affected largely by the physical conditions of the site, chiefly soil type and weather (see Figure III-20). The two latter categories are what can be called the site-conditions, even though only the physical conditions of the site have been usually considered when talking about site-dependency. Thus, it is suggested here that the concept "site-dependency" should have a broader sense, in order to include not only the actual characteristics of the physical site, but also the habits of the producer that is using that site (techniques are chiefly shaped by technology type, but affected to a great extent by human habits). In Figure III-20, thus, the final environmental interventions (the results of the inventory analysis) are shown as being determined first by the technology type, which gives information on the type of substances being used; then, the amounts and specific substances are generally decided by the farmer (technique), and the predicted emissions highly depend on site conditions.

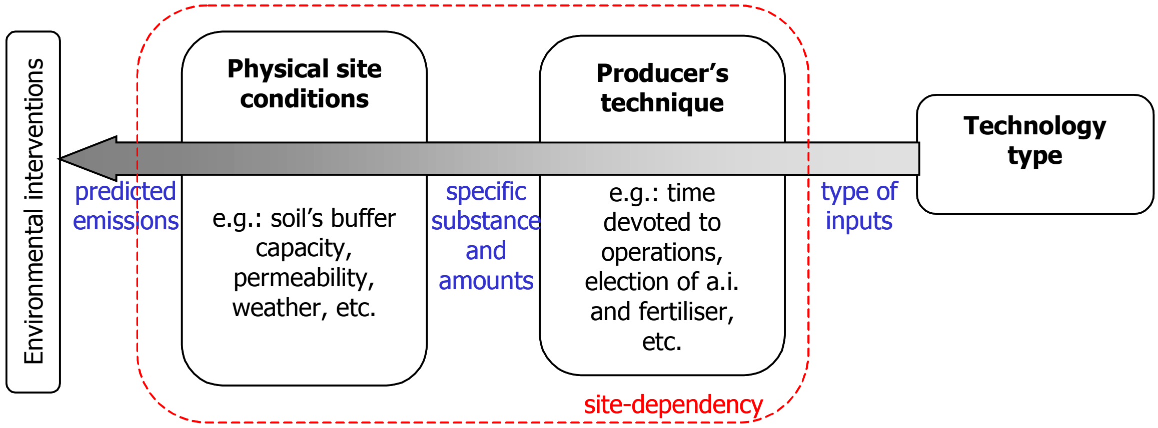

Figure III-20: Aspects influencing the LCI results.

The interpretation in Table III-42 (page 129) suggests that site-dependency in the systems' contribution to impact categories is more frequent in IFP than in OFP. This is due in some occasions to the lower uncertainty considered for IFP systems (see the lower uncertainty ranges for eco-toxicity through water in IFP systems in Table III-38, page 124). In most cases, though, it is because the aspects mostly determining some of these impact categories are affected by farmer's practices to a greater extent in IFP: election of active ingredient and fertiliser type determining human toxicity through water and soil, and acidification (Figure III-20). The effects of site-conditions (both of the physical conditions and the producer's practices) on the inventory results have thus been found to have a greater influence on the LCA results than activity-dependent aspects for some impact categories and are further discussed in the following paragraphs.

Producer's practices (technique)

Different farmer's practices lead to highly variable results. This is due to different machinery use and efficiency in different orchards, different agro-chemical and fertiliser dosage, practices for agro-chemical application, and irrigation patterns. The consequences of use of different techniques within the same technology are highlighted in Table III-44.

Table III-44: Main technique-dependent aspects from the LCI that affect the LCA results.

Technique-dependent aspect	*Reason / Observations*
Degree of mechanisation	Machinery use (in hours/ha) devoted to each field operation varies enormously between orchards (see Table III-14, in page 68), and this has a direct effect on the contribution to several impact categories through energy consumption and, to a lesser extent, machinery production. Also the type of machinery used (Table III-13, page 66) presents wide variations and specific fuel consumption (Table III-26, page 89) may vary due to the machine's engine rate and driver's habits.
Efficiency in machinery use	As machinery production plays and important role on the results, which determines differences in contributions by different producers, the intensity of use of each machine should be included in the LCI. Thence, if a machine is used very intensively, the impacts per hour of use are smaller (the total use in lifetime is longer, see Table III-25 in page 88). This effect could not be included in the analysis, because no data was available on total use per each machine, but only on machine use per operation and theoretical total machine use (from the literature).
Choice of active ingredients for Pest & Disease Management	This has a direct effect on the LCI and LCIA results, through the physical-chemical properties of the substance (which determines fate, see III.2.4) and its intrinsic toxicity (which determines the effect, see Table III-33). Today, there is a wide array of active ingredients that can be used for a specific reason, and thus this election should be done considering the environmental effects. Of the physical-chemical parameters, mainly the volatility and biodegradation half-life have been concluded to be determinant for the LCI results (see section III.3.3).
Timing of pesticide applications	As it was explained in section III.2.4, timing of pesticide application has a direct effect on spray deposition (due to the presence/absence of foliage). Also the incidence of rainfall on pesticide leaching determined by PESTRISK depends on the time when pesticides are applied. Finally, early applications of pesticides allow for a longer biodegradation period, thus reducing emissions to soil. Of course, the farmer does not arbitrarily decide timing of pesticide applications, but weather often dictates the possible dates for spraying, and pests' life cycles pose different needs in different times of the year.
Pesticide spraying technique	Also the way in which the pesticides are applied has a direct effect on the deposition. Mainly whether spraying is done at dilute or concentrate volumes (which is an election of the farmer) determines this effect.
Choice of fertiliser	The election of the fertiliser also has clear effects on the results, as different fertilisers have highly variable specific emissions in field. In general, nutrient management should be regarded from a very site-dependent point of view. Therefore, not only fertiliser characteristics but also site conditions should be taken into account to perform a nutrient balance that allowed for a consistent prediction of nutrient field emissions. Also timing and application methods should be considered in this respect.
Irrigation technique	Depending on the irrigation practices, more pesticides and nutrients may be prone to leaching. Irrigation patterns show wide variations between locations, suggesting that not much agronomic foundation is used when deciding the amount of water and timing of irrigation. Also the technology used for irrigation has some effect on the final results, through the influence on energy consumption. Nevertheless, as most energy used for irrigation is electricity, and in New Zealand this comes from hydroelectric plants (with low environmental impacts), no evidence is shown in the results.

Physical conditions of the site

Not only the producer's practices (technique) affect field emissions. As it has been pointed in the results (section III.3.2), similar inputs may result in very different emissions due to the physical conditions of the site. In the New Zealand apple LCA this fact was addressed in the fate analysis of

pesticides, mainly through soil characteristics and weather conditions, which are included in the PESTRISK model. For example, similar use of the active ingredients cyprodinil and carbaryl in the IFP orchards give a much higher contribution to Human Toxicity through Water in the Bennies' orchard (IFP_CO_1), due to its soil type. The detailed effects of soil properties or weather patterns on pesticide leaching are addressed in HortResearch (2000), where soil organic carbon and water recharge rate are pointed as the two key soil-dependent parameters affecting pesticide leaching.
The physical conditions of the site may also affect the environmental impacts through the orography. Even though no significant differences were found in the apple LCA because all farms had similar orography, it may be argued that a tractor operating within a farm with steep slopes might consume more energy. Also runoff and thus pesticide and fertilisers emissions will be affected by field's slope. The effect of orography has only been included through the different sources of water, which imply different energy consumption for its delivery.

Relation to the findings of other authors

Other authors have suggested that in agricultural LCA the choice of location is a valid difference between systems, because site-dependent aspects can have greater influence on LCA results than technology-dependent aspects (Cowell & Clift 1998). Cowell (1998) mentions climate and soil type as the two main factors determining e.g. the yield in an agricultural system, and thus the product to refer the impacts to. Beaufoy (2001) related the variations found in olive farming within the EU across three broad categories:

- plantation characteristics and farming practices (referring to technology type –organic, integrated…-, size and distribution of the plantation, and producer's habits),
- physical and biological conditions in which farming takes place (site conditions, including soil and weather, as well as type of habitats in relation to the impacts on biodiversity),
- the socio-economic situation of the holding (which determines things such as the expenditure on labour and/or machinery).

While the effects of the two first groups have been detected in Table III-42 and Figure III-20, and generally discussed in the last paragraphs, the socio-economic conditions of the different orchards under study were not analysed[25]. As it has been pointed out throughout the chapter, organic apple production in New Zealand is highly mechanised. Thus, even though it is called "organic" because of its avoidance of particular inputs, it has nothing to do with the idea of "extensive" agriculture that is sometimes evoked by that name (see the semantic discussion in section III.4.3, sub-section "Criteria for organic agriculture"). The environmental and economical effects of substituting labour for machinery could not be explored in the apple LCA, although social, economic and environmental issues would rise in such a discussion. Obviously, the total economic costs of apple production would increase if hydra-ladder use had to be reduced at the expense of (expensive) human labour. The environmental effects of hand-made operations (chiefly thinning, pruning, harvesting) would then be reduced, but the total environmental burdens should be allocated among less (economic) output even in the case that yield did not vary (considering net benefits, rather

[25] It seems obvious, though, that they will affect the use of machinery and the energy consumption, as the producers will seek to maximise benefits by reducing costs of inputs while keeping a balance with yield.

than gross sale value, as the economic output). In a poorer country, on the other hand, probably the same substitution of labour for machinery would render reduced environmental effects without increase in economic costs. Indeed, in a poorer country agriculture would naturally be less machinery-intensive.

This is reflected in Figure III-21, which is adapted from Figure III-20. Here the socio-economical situation of the farm is depicted as another factor indirectly determining the environmental impacts, mainly through its effects on the producer's practices. Chiefly, and as discussed above, the relation between labour and fuel (mechanisation) costs will determine whether any operation is performed in a more labour- or machinery-intensive way.

Other socio-economical aspects, such as the expected profit with each technology (from the retail value of products), policy, subsidies, etc, will affect the election of technology. Obviously, also know-how from the producer and personal preference will also be key. Indeed, it is an interesting perception that some of the organic producers did not choose to produce organic as a way of making money, but as an election of life-style. This might actually be part of the explanation for the high energy consumption found in some OFP orchards: the producers are not so much concerned by economic profit, and thus do not work so hard on energy-savings.

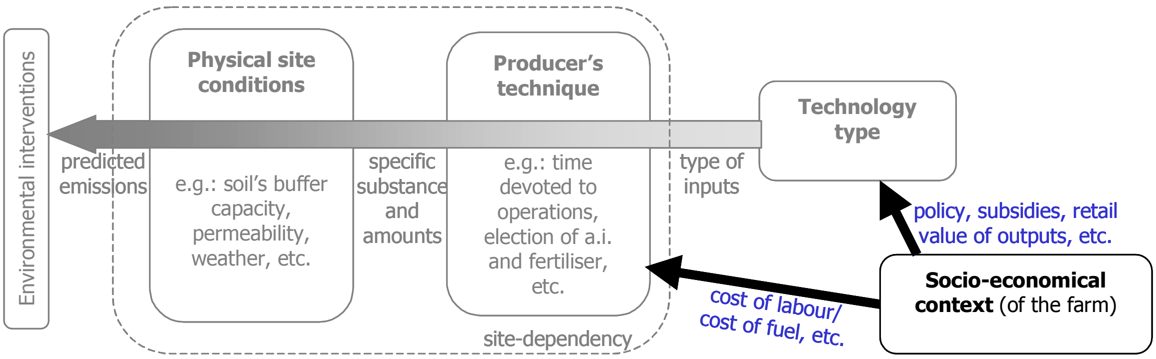

Figure III-21: Indirect effects of the socio-economical context on the LCI results (environmental interventions).

As a general conclusion, it should be stated that site-dependent conditions should be included in LCA, and particularly in agricultural LCA. Not only the "common" site-dependent conditions such as soil type and climate must be assessed, but also particular farmer's techniques have been found to be key in determining the overall impacts of agricultural systems in the New Zealand apple LCA.

III.5.2. Site-dependency in impact assessment

The effects of site-dependency (both the physical conditions of the site and the farmer's techniques) on LCIA were not sufficiently explored in the New Zealand apple LCA. Actually, it is still not clear how site-dependency should be included in LCIA, though the answer to this question is likely to depend on the impact category. Indeed, some authors (Cowell 1998; Haas *et a*l. 2000) have already suggested that some impact categories should be treated as site-independent, while other categories may have increasing degrees of site-dependency. This is also implicit in Consoli *et al*. (1993) through the recognition that the effects of different environmental impact categories may be relevant at different levels: global, continental, regional and/or local. All these categories, and their consequences for the New Zealand apple LCA, are discussed below.

Site-independent impact categories

The impact categories traditionally considered being site-independent are global warming, ozone depletion, and both biotic and abiotic resource depletion. The argument to keep these impact categories as site-independent is that they have global rather than regional or local impacts. This argument can at least be partially refuted in the case of the abiotic resource water, where the relevance of considering regional indicators has already been addressed in the literature (Lindeijer *et al*. 2002; see also Cowel 1998).

Water is indeed a very special case: as a resource it is considered either a flow or a fund, and cannot be actually depleted, but what is at stake in this case is its availability and quality. Resource availability is also the key point in all other resources, but as they are commonly transported when needed, local availability is not that relevant. Water, so far, needs to be locally available for it to be considered (technically) available. Therefore, site-conditions should be included in the characterisation process for the impact category to be more credible. Besides, water has a multi-functional nature, and thus impacts related to its depletion should be treated accurately. Water may indeed affect many other impact categories: toxicity (it is a vehicle for contaminants), soil erosion (vehicle for soil particles), land productivity (water is key for life) and thus land's life support functions and biodiversity, etc. Indeed, water use may be more linked to land use impacts than to abiotic resource depletion, and thus should be considered from a site-dependent point of view (see below).

As an example, the case of apple production in New Zealand may be mentioned. In Hawke's Bay, groundwater is pumped for irrigation and frost fighting, while surface water is used in Central Otago. From the resource point of view, the categories of water used are thus different, and even though groundwater depletion is still not a serious issue in New Zealand in the two regions studied, its consumption is of greater concern than that of surface water. Technically, groundwater is considered as a fund (it is renewed, but the rate of renewal should not be surpassed), while surface water is a flow (it cannot be depleted, only competition for its use is relevant). Characterisation factors should thus be developed including local conditions (water scarcity and aquifers' renewal rate in different New Zealand regions), and water depletion be included as an impact category in the LCA if it was found to be relevant for the decision-makers.

On the other hand, also biotic resource depletion may have regional effects. A clear example is the local loss of a species, affecting local biodiversity (biodiversity on a global scale might not be affected, but the loss at local level might have further consequences on related species). Abiotic resource depletion and biodiversity could neither be included in the apple LCA.

Categories that should include some degree of site dependency

A second group of impact categories should include some degree of site-dependency, both in industrial and agricultural LCA. Amongst the categories analysed in the apple LCA, these include toxicity related categories, nutrification and acidification, and photochemical oxidants formation. The effects of these categories may be relevant from the global to the local scale, according to Consoli *et al*. (1993).

In the case of toxicity categories, in some occasions the proximity of human populations or sensitive ecosystems to the point of release of toxic substances is known. In these cases, acute effects of short-lived toxic substances should be included (which is done to a certain extent in the acute effects of ecotoxicity through water). In fruit production, some cases are known of intoxication from pesticides in orchard's neighbourhoods while spraying in windy days.

The categories of nutrification and acidification have always been regarded as having regional effects. Pujol & Boidot Forget (1994) suggest including some site-specific information in the LCI in order to consider it during the interpretation of impacts on nutrification. José Potting (2000) developed a site-dependent approach that allows for more credible results of these categories when characterising European emissions. Nevertheless, data were not available for New Zealand conditions, and thus site-dependency was not included for these categories in the apple LCA. The lack of such a model for photochemical oxidants formation made it impossible to even consider site-dependency in this impact category, even though some preliminary approaches have been made to distinguish between emissions occurring in areas with high or low NO_x concentrations (see e.g.: Hauschild & Wenzel 1998).

Site-dependent impact categories: Land use related impacts

Finally, there is a group of impact categories that is site-dependent almost by definition. This is particularly the case of land use derived impacts and local impacts derived from the geographical distribution of different land uses, such as "nuisances" (visual, acoustic, and odour contamination) and radiations. Land use impact categories cover damage on biodiversity (depending on local biodiversity), soil quality or land fertility (depending on initial soil conditions), and land competition (depending on local availability of land). As impacts from land use are usually expressed as the difference between a current state and a potential (relaxation) state (Lindeijer *et al*. 2002), and this potential depends clearly on the local (site) conditions, this category should be treated as site-dependent, even though the effects of land degradation are of global concern.

In the case of agriculture, some of these aspects gain a special relevance. Local biodiversity is also important from a functional point of view, because the presence/absence/abundance of pest predators affects the need for pest management (Suckling *et al*. 1999). Soil quality has been an extensive matter of research, and it seems to be a key difference between IFP and OFP; to date, many different indicators have been studied showing this (Daly *et al*. 1996; Marsh *et al*. 1996; Hartley & Rahman 1997; Marsh *et al*. 1998). Last, but not least, land competition is very relevant especially in New Zealand because productivity is a crucial issue in a country so aimed at exportation; of course, this issue lies more within the economical sphere than within the environmental issues.

In studies where land use impacts are suspected to be important, such as in agricultural LCA, it may be necessary to make a thorough investigation of the issue (Mattsson *et al*. 2000). Nevertheless, no clear and operational methods exist so far for land use related impacts, and these are regarded as being crucial for the credibility of agricultural LCA. Particularly the study of soil quality and its inclusion in LCA studies is considered key, and encouraged the work on an indicator for soil quality impacts to be used within the framework of LCIA. Soil quality and its inclusion in LCIA is the object of study in Chapter IV of this thesis.

III.6. Conclusions for site-dependency in agricultural LCA

The application of the state-of-the-art LCA methodology to an agricultural system has lead to several important conclusions:

- The results are highly dependent on the characteristics of the site. Soil and weather had already been highlighted in the literature as key factors affecting the impacts in agricultural systems and not very much in industrial systems, which represents an important difference between these systems. On the other hand, the extent of the effect of different farmer's practices on the results had not been properly addressed in agricultural LCA. In this thesis, it has been shown how farmer's practices determine the amount of resources consumption, and thus indirectly the amount of emissions from the system.
- Soil and weather (the physical site conditions) act as "filters" reducing or increasing emissions to the environment from the amounts used by the farmer. As physical site conditions, and above all farmer's practices, may vary largely from site to site, it is highly relevant to consider these differences when comparing agricultural systems by means of LCA.
- Some differences in farmer's practices (e.g.: variations in pesticide use) might level out when whole crop rotations are analysed as is suggested for agricultural LCA. On the other hand, some other differences are likely to increase (e.g.: fuel consumption). This should be further explored in future applications of agricultural LCA.
- Site characteristics have been found to affect the LCA results to a bigger extent than the choice of technology (organic or integrated) in many impact categories. Actually, only the impact categories that are clearly affected by substances only used in IFP (synthetic pesticides) show clear differences between IFP and OFP (Human Toxicity through water and soil). Also those impact categories chiefly dominated by energy consumption (Eco-Toxicity through soil and Photochemical Oxidants Formation) present clear differences, because in the New Zealand apple LCA a consistenly higher energy consumption has been found for organic orchards.
- The intensive mechanisation of field operations has been found to seriously hamper the environmental preference of OFP over IFP in New Zealand. Therefore, it has been here shown that if organic agriculture is to actively contribute to bridging the gap between current practices and sustainable food (and fibre, and timber…) production, new indicators such as the level of mechanisation should be taken into account.
- The complexities of agricultural systems due to their closeness with nature, and the necessity of integrating site-conditions in the assessment of environmental impacts, require a trans-disciplinary approach for the completion of agricultural LCA. Particularly when LCA is to be applied for the assessment of good agricultural practices (GAP), it is here suggested that close cooperation with agronomists and environmental modellers is necessary, in order to be able to respond to the needs for system knowledge.
- Many opportunities for improvement have been found for apple production in New Zealand thanks to the application of LCA. In the cases where these opportunities might seem

obvious even before any LCA application, the relevance of their application has been shown. Therefore, decisions on technology choice can now be made based on solid information.

- Most improvement opportunities require the cooperation from farmers. The LCA results and opportunities for improvement should thus be communicated to them connecting the environmental requirements with their socio-economical needs, in order to get proactive attitudes. Environmental improvement usually means reduced costs for inputs (energy, pesticides) and thus result in “win-win situations”; the difficulty in communicating such improvements lies in the fact that farmer’s habits are usually shaped by pre-conceptions of “what has always been done”, which is difficult to change.
- Machinery production is responsible for a relevant share of energy consumption in the apple LCA (around 7-15% of total energy consumption), thus confirming previous studies. In relation to this, one of the biggest data gaps that have been found is related to the machinery use. Some research effort is needed to produce high-quality data for machinery production, and particularly for the conditions of machinery use. The total amount of hours a machine is used over its lifetime largely determines the environmental consequences of its use, and therefore data of high quality is needed.

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APPENDIX III.1. Questionnaire template

The questionnaire sent to the apple growers during May and June 2000 by HortResearch is provided in the next pages. As it has been noted before, the questionnaire was first filled by the growers, and then many phone calls were used (July and August 2000) in order to check the information and fill any data gap.

Apple LCA

A request for information for Life Cycle Assessment of apples

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Lincoln

Tel 03 325 6602
Fax 03 325 6063
Email gburnip@hort.cri.nz

Participant

Name:____

Orchard name:____________________________________

Postal address:___________________________________

____

____

Tel: ________________________

Fax:________________________

Email:_______________________

Machinery Table

Please list all machinery used on your orchard[1]. Machinery is defined as any equipment that use the fuel types/energy sources listed below[2]. Use the assigned machinery number (column 1) to identify the use of the machinery in the following sections.

[1]Please provide all descriptive names/numbers used by the manufacturer to identify the model.

[2]Fuel type/energy sources = diesel, petrol (91 or 96), gas (state type), batteries, electricity

[3] cc rating for engines; Pumps: flow rate (litres/sec), pump power (kW)

Machinery number	Description	Make	Model[1]	Rating[3]	Fuel/energy type[2]
1					
2					
3					
4					
5					
6					
7					
8					
9					
10					
11					

Machinery number	Description	Make	Model[1]	Rating[3]	Fuel/energy type[2]
12					
13					
14					
15					
16					
17					
18					
19					
20					
21					
22					
23					
24					
25					

Machinery check list:
Tractors
Forklifts
Mowers
Sprayers
Fertiliser spreaders
Mulch spreaders
Mulching mowers
Hydra-ladders
Motorised pruning systems
Irrigation/frost fighting pumps
Wind machines
Smoke pots
Diesel or propane burning orchard heaters

Orchard data
Size of orchard (planted area)
cultivar mix
tree / row spacings

General comments regarding recording machinery use
In the information you provide below on machinery use, please be mindful that what is required is an estimate of the time that the machinery was used to accomplish a task (per unit of area). If the machinery is operated for periods not directly associated with accomplishing a task in a given area, but is still necessary for this task , a proportion of this indirect machinery use should be apportioned.

For example, the time taken to fill an airblast sprayer (with tractor running), drive to a block and return, should be apportioned to the tractor use time associated with spraying a one ha. block. If two one ha. blocks are able to be sprayed with one tank (and one trip back & forth to the water source/spray shed) then half this time should be apportioned to the one ha. block under consideration.

Another example might be a wind machine which is required to be operated for ½ hour, every 2 months, to maintain operational status – this is a maintenance cost of the equipment which should be factored into the seasonal use of the wind machine. Such apportioned costs (time, energy use etc) are only a consideration when they are likely to add significantly (greater than 10%) to the overall value. Please use your judgement on this issue.

Mowing
Machinery used (enter machinery number from table above) ______________________________

Time (hours/minutes) ___________ taken to mow _______________ area (acres / hectares) of orchard

Number of times this area is mown per season. _________________________

Airblast Spraying
Machinery used (enter machinery number from table above) ______________________________

For a "typical or average" block (or entire orchard), record the
Time (hours/minutes) ____________ taken to spray this block (or entire orchard).

Size of this block (or entire orchard) ____________ (acres / hectares).

Number of times this block (or entire orchard) is sprayed per season ________________________
i.e. number of passes per season.

Herbicide Spraying
Machinery used (enter machinery number from table above) ______________________________

For a "typical or average" block (or entire orchard), record the
Time (hours/minutes) ____________ taken to spray this block (or entire orchard).

Size of this block (or entire orchard) ____________ (acres / hectares).

Number of times this block (or entire orchard) is sprayed per season ________________________
i.e. number of passes per season.

Weed-eater use
Machinery used (enter machinery number from table above) ______________________________

For a "typical or average" block (or entire orchard), record the
Time (hours/minutes) ____________ that a weed-eater is used within this block (or entire orchard).

Size of this block (or entire orchard) ____________ (acres / hectares).

Number of times weed-eater is used wihtin this block (or entire orchard) ____________________

Machinery assisted pruning
Machinery used (enter machinery number from table above) ______________________________

For a "typical or average" block (or entire orchard), record the time (hours/minutes) ____________
taken to prune this block (or entire orchard), using the above machinery. Please only record the time that machinery is in use, ie do not record time involved in non-motorised hand pruning.

Size of this block (or entire orchard) ____________ (acres / hectares).

Disposal of prunings
Machinery used (enter machinery number from table above) ______________________________

For a "typical or average" block (or entire orchard), record the time (hours/minutes) ____________
taken to dispose of the prunings generated from pruning this block (or entire orchard), using the above machinery.

Fruit thinning sprays
Machinery used (enter machinery number from table above) ____________________

For a "typical or average" block (or entire orchard), record the time (hours/minutes) __________ taken to apply thinning sprays to this block (or entire orchard).

Size of this block (or entire orchard) __________ (acres / hectares).

Number of times this block (or entire orchard) is sprayed per season ____________________ i.e. number of passes per season.

Hand thinning
Machinery used, e.g. hydra-ladders (enter machinery number from table above) __________

For a "typical or average" block (or entire orchard), record the time (hours/minutes) __________ this machinery is in use within this block (or entire orchard), during the season.

Size of this block (or entire orchard) __________ (acres / hectares).

Number of times per season this machinery is used for thinning in this block (or entire orchard) ____________________.

Irrigation
General description of irrigation system ____________________

____

____

Water source: surface water (river, stream, lake) or groundwater (bore, artesian)__________

____

As the irrigation system of each orchard will be different, I may need to contact you further on this aspect.

Machinery used, (enter machinery number from table above) __________

For a "typical or average" block (or entire orchard), record the time (hours/minutes) __________ irrigation machinery is in use within this block (or entire orchard), during the season.

Size of this block (or entire orchard) __________ (acres / hectares).

OR If you know the amount of electricity used to irrigate a known area of your orchard, please record this here (kW* / ha)____________________

* a meter reading of 100 units = 100kW

How often is irrigation applied (on a block or total orchard basis)______________________

____

Frost fighting
General description of frost fighting system ______________________________________

____

____

Machinery used, (enter machinery number from table above) ____________

For a "typical or average" block (or entire orchard), record the time (hours/minutes) __________ this machinery is in use within this block (or entire orchard), during the season.

Size of this block (or entire orchard) __________ (acres / hectares).

How often is frost fighting carried out (on a block or total orchard basis)_______________

____

Harvesting
Machinery used, (hydra-ladders?) (enter machinery number from table above) ____________

For a "typical or average" block (or entire orchard), record the time (hours/minutes) __________ this machinery is in use within this block (or entire orchard), during the season.

Size of this block (or entire orchard) __________ (acres / hectares).

Transport of fruit within the orchard
This should include the transport of fruit (normally in bins) from harvested blocks to a central location, prior to cool-storage, trucking, or grading.

Machinery used, (enter machinery number from table above) ____________

For a "typical or average" block (or entire orchard), record the time (hours/minutes) __________ this machinery is in use within this block (or entire orchard), during the season.

Size of this block (or entire orchard) __________ (acres / hectares).

Fertiliser application
Machinery used, (enter machinery number from table above) ____________

For a "typical or average" block (or entire orchard), record the time (hours/minutes) __________ this machinery is in use within this block (or entire orchard), during the season.

Size of this block (or entire orchard) ___________ (acres / hectares).

Mulch application
Machinery used, (enter machinery number from table above) _______________

For a "typical or average" block (or entire orchard), record the time (hours/minutes) ___________ this machinery is in use within this block (or entire orchard), during the season.

Size of this block (or entire orchard) ___________ (acres / hectares).

If there are other practices that are not covered above, can you please provide details in the above format.

Soil analysis
Can you please supply copies of any soil analysis reports you have from your orchard for the previous 2 seasons.

Spray diaries
Can you please supply copies of your spray diaries from your orchard for the previous 2 seasons. If you do not have these readily at hand, I can probably access them from ENZA internally.

The data needed for the calculations of pesticide fractions reaching each compartment (see section III.2.4) is extensive, and usually hard to get. Many data from toxicity studies is also needed for the calculation of human- and eco-toxicity characterisation factors (Hauschild & Wenzel 1998). The fact that most pesticides are continuously improved and the active ingredients change very often explains most of this difficulty. Besides, all data should come from the same source, as this improves considerably the consistency of the modelling, but further complicates the research.

Main pesticide data sources

Probably the best data source that can be found for pesticide modelling and toxicity data is The Pesticide Manual, which currently incorporates the Agrochemicals Handbook (Tomlin 1995). At least, this is the publication from where most of the data in this dissertation come, and it is the only one I have found covering absolutely all of the active ingredients involved in the apple LCA (even those for OFP). Nevertheless, not all data could be found in this exhaustive book, and so other sources had to be consulted. Today, most of these sources can be consulted in Internet, which has the advantage of almost permanent update. There are many useful websites that I had to use in order to gather all the data, and in Table III-45 I organise them according to their main strength: physical-chemical data for fraction modelling or toxicity (both human and ecological) data.

Material Safety Data Sheets (MSDS)[26] from the pesticide manufacturer have also been used in a number of cases, as they represent an easily accessible source of data with many characteristics relevant for the modelling, and particularly for toxicity values. Properties such as the vapour pressure, solubility (in water and other substances) and partition coefficients can be found in a MSDS. Apart from Reference Doses and ADI[27] for humans, modern MSDS usually include (acute) eco-toxicity data for the different organisms required by Hauschild & Wenzel (1998) for a sound calculation of effect factors: fish, Crustacea and algae.

[26] A Material Safety Data Sheet (MSDS) is a document that contains information on the potential health effects of exposure and how to work safely with the chemical product. It contains hazard evaluations on the use, storage, handling and emergency procedures all related to that material. The MSDS contains much more information about the material than the label and it is prepared by the supplier. It is intended to tell what the hazards of the product are, how to use the product safely, what to expect if the recommendations are not followed, what to do if accidents occur, how to recognize symptoms of overexposure, and what to do if such incidents occur. (http://www.ccohs.ca/oshanswers/legisl/msdss.html [on-line: 18.07.2002])

[27] Acceptable Daily Intake. This is an estimate of the amount of a pesticide in food and drinking water which can be ingested daily over a lifetime by humans without appreciable health risk. It is usually expressed in milligrams per kilogram of body weight.

Table III-45: Main Internet sources for pesticide physical, chemical and toxicological data.

Sources for physical-chemical pesticide data	
website	**reference, main data, updates**
http://toxnet.nlm.nih.gov/cgi-bin/sis/htmlgen?HSDB	**HSDB**. Hazardous Substances Data bank (HSDB). An exhaustive compilation of references dealing with a wide array of toxicity related issues is found in this website. Human health effects, emergency treatments, pharmacology, environmental fate and exposure, standards and regulation, physical-chemical properties, safety & handling, etc. of many hazardous substances (not only pesticides) are addressed. In the TOXNET homepage (http://toxnet.nlm.nih.gov/) other interesting data sources can be found. Currently (2002) updated.
http://esc.syrres.com/efdb.htm	**EFDB**. Environmental Fate Data Base. Different databases can be found in this website, mainly dealing with the environmental fate of substances: biodegradation, physical-chemical properties, etc. A particularly interesting one is CHEMFATE, a data value file containing 25 categories of environmental fate and physical/chemical property information on commercially important chemical compounds. Currently (2002) updated.
http://ace.ace.orst.edu/info/extoxnet/pips/ghindex.html	**PIP**. EXTOXNET, the EXtension TOXicology NETwork. Thorough pesticide descriptions, the so-called Pesticide Information Profiles (PIP), are found in this website. Information on manufacturer, toxicological and ecological effects, environmental fate, exposure guidelines, and physical properties is given for a number of pesticides. No information on last update is given, although I have not seen any change in it since 2000.
http://wizard.arsusda.gov/acsl/ppdb.html	**ARSPPDB**. The ARS (Alternate Crops & Systems Lab.) Pesticide Properties Database. Mainly physical-chemical properties (vapour pressure, solubility in water and organic solvents, partition coefficients, rate constants, half-lives, etc.) are found here, with an extensive list of pesticides (324). No information on updates is given.
http://www.msdssearch.com/DBLinksN.htm	This web page contains links with the main MSDS databases, which facilitates the search of a MSDS for a particular compound. Currently (2002) updated.
http://www.mtas.es/insht/ipcsnspn/spanish.htm	Access to the International Chemical Safety Cards (ICSC) for most chemicals, in Spanish. Information on some chemical properties is included, but no toxicity values, apart from exposure limits, are available. Not updated since November 1999.
Sources for toxicological pesticide data	
website	**reference, main data, updates**
http://www.pesticideinfo.org	**PAN**. The Pesticide Action Network (PAN) Pesticide Database offers current toxicity and regulatory information for pesticides. It brings together a diverse array of information on pesticides from many different sources, providing human toxicity (chronic and acute), eco-toxicity and regulatory information for about 6,000 pesticide active ingredients registered for use in the United States. Also information on their transformation products, as well as adjuvants and solvents used in pesticide products, is provided. In some cases, also information on properties affecting the environmental fate of pesticides is provided. Many links to other powerful data sources are provided. Currently (2002) updated.
http://www.epa.gov/ecotox/	Site of the US EPA Database on eco-toxicity, including the Aquire database. It provides single chemical toxicity information for aquatic and terrestrial life. Peer-reviewed literature is the primary source of information encoded in the database. Currently (2002) updated.

Extrapolation of data

Obviously, not all data can always be found, and in some cases extrapolations for some values have to be made. In these cases, I have followed a procedure similar to the one used for pesticide production data (suggested by Audsley *et al*. 1997, as mentioned in section III.2.6). Thence, when some figure was missing for a given pesticide, I have used the average of other pesticides of the same chemical group. In case there was no data for other pesticides in that chemical group, average data for the agrochemical type (herbicide, insecticide, fungicide, or plant growth regulator) were used. The consequences of this procedure for each pesticide can be found in Table III-46's footnotes.

Table III-46: Main pesticide data for the determination of the fractions of pesticide entering different compartments.

Pesticide [a]	Active Ingredient		CAS # [b]	Chemical group	Main [c] manufacturer	Vapour Pressure (Pa)	Water solubility (mg L^{-1})	Half-life in soil (days) [d]	Half-life on the plant (days) [e]	log K_{OW}	K_{OC}
Alsystin 25WP	Triflumuron	25%	064628-44-0	Benzoylurea (Ecdysteroid agonist)	Bayer	4.00E-08	0.025	112.0	34.4 [i]	4.91	4150.33 [i]
Amitrole 400	Amitrole	40%	000061-82-5	Amino triazole	Bayer	5.50E-08	280	3.0	19.8	-0.65	93.00
Applaud 25W	Buprofezin	25%	069327-76-0	Thiadiazine	Syngenta	1.25E-03	0.005	30.0	4.6 [ii]	4.30	92.98
Basudin 50WP	Diazinon	50%	000333-41-5	Organo-phosphate	Ciba-Geigy	1.20E-02	60	16.0	4.0	3.30	1581.00
Bayleton 5DF	Triadimefon (Triadimenol)	5%	043121-43-3	Triazole	Bayer	2.00E-05	260	12.0	0.6	3.12	300.00
Captan 80WP	Captan	80%	000133-06-2	Cyclic imide	Zeneca	1.30E-03	3.3	1.0	0.0	2.79	198.00
Chorus	Cyprodinil (CGA219417)	50%	121552-61-2	Anylinopyrimidine	Syngenta	5.10E-04	13	40.0	20.0 [iii]	4.00	1470.00
Delan WG	Dithianon	70%	003347-22-6	Quinone	Cyanamid	2.71E-09	0.14	1.0	4.6 [ii]	2.84	893.60 [ii]
Dodine 400	Dodine (Cyprex)	40%	002439-10-3	Guanidine derivative (substituted acetate)	Cyanamid	2.00E-05	630	20.0	4.6 [ii]	-0.88	10000.00
Fruitfed ANA 10%	1-Naphtyl Acetic Acid	10%	000086-87-3	Synthetic auxin	Rhône-Poulenc	1.00E-02	420	10.0	1.9 [iv]	2.60	200.00
Gusathion M-35	Azinphosmethyl	35%	000086-50-0	Organo-phosphate	Bayer	1.80E-04	28	10.0	99.0	2.96	940.00
Lorsban 50EC	Chlorpyrifos	50%	002921-88-2	Organo-phosphate	Dow	2.70E-03	1.4	90.0	0.2	4.70	9930.00
Manzate 200	Mancozeb	75%	008018-01-7	Dithiocarbamate	DuPont	1.31E-05	13	10.5	0.1	1.33	2000.00
Match	Lufenuron	5%	103055-07-8	Benzoylurea	Syngenta	4.00E-06	0.06	16.0	34.4 [i]	5.12	4150.33 [v]
Mimic 70W	Tebufenozide	70%	121410-23-8	Diacylhydrazine (Ecdysteroid agonist)	Rohm&Haas	3.00E-06	0.83	66.0 [iii]	34.4 [i]	4.25	605.00
Mizar Granuflo	Ziram	76%	000137-30-4	Dithiocarbamate	Bayer	1.00E-06	0.03	30.0	0.1 [vi]	1.09	400.00
Nimrod 25WP	Bupirimate	25%	041483-43-6	Pyrimidine	Zeneca	1.00E-04	22	60.0	4.6 [ii]	2.70	893.60 [ii]
Nustar 20DF	Flusilazole	20%	085509-19-9	Triazole	DuPont	3.90E-05	49.5	420.0	3.0 [vii]	3.74	1500.00
Pallitop	Nitrothal Isopropil	50%	010552-74-6	Nitrogen compound	BASF (pallinal)	1.00E-05	0.39	6.0	4.6 [ii]	2.04	893.60 [ii]
Polyram DF	Metiram	70%	009006-42-2	Dithiocarbamate	BASF	1.00E-05	1	20.0	0.1 [vi]	0.30	100000.00
Roundup G2	Glyphosate	36%	001071-83-6	N-(phosphonomethyl) glycine: Organo-phosphate	Monsanto	3.85E-08	12000	41.0	35.0	-4.00	2100.00

Score 10WG	Difenoconazole	10%	119446-68-3	Triazole	Syngenta	3.30E-08	16	420.0 [viii]	6.0 [iii]	4.20	2000.00 [iii]
Sevin WP	Carbaryl	80%	000063-25-2	Carbamate	Rhône-Poulenc	4.10E-05	120	15.0	1.9	1.59	288.00
Stroby WG	Kresoxim Methyl (BAS490F)	50%	143390-89-0	Strobilurin	BASF	2.30E-06	2	1.0	4.6 [ii]	3.40	437.00
Syllit Plus	Dodine	40%	002439-10-3	Guanidine derivative	Cyanamid	2.00E-05	630	20.0	4.6 [ii]	-0.88	10000.00
Systhane 40W	Myclobutanil	40%	088671-89-0	Triazole	Rohm&Haas	2.13E-04	142	66.0	2.3	2.94	500.00

SOURCE: Several data sources have been used in order to fill this table, as is explained in the text. The main ones are Tomlin (1995), HSDB, PAN, PIP, EFDB and ARSPPDB (see the full references in Table III-45, in page 164). The figures not present in these references have been extrapolated using the procedure explained above, and this has been properly explained in a footnote numbered in Roman numerals.

[a]: The pesticide commercial names reported by the apple producers participating in the study are reproduced here. In some cases, the same active ingredient may be sold using several commercial names, and even by different manufacturers.

[b]: Chemical Abstracts Reference Number.

[c]: The same active ingredients may be produced by more than one manufacturer. Usually, the manufacturer of the commercial product used in the study is reproduced here. It must be noted that most chemical companies in the world are under a process of growth and merging. For example, Syngenta is a new company resulting from the fusion of several agrochemical companies: Novartis, Sandoz, Ciba-Geigy, Nihom Nohyaku and possibly some others; Dow Chemical recently acquired former Rohm & Haas; etc. As most data sources are older than 2 or 3 years, the manufacturer's name might be outdated in some of the substances.

[d]: Usually determined by the rate of microbial degradation (Hauschild 2000).

[e]: Usually determined by the rate of photolysis (Hauschild 2000).

[i]: as average insecticide (Diazinon, Azinphosmethyl and Chlorpyrifos).

[ii]: as average fungicide (Triadimefon, Captan, Cyprodinil, Mancozeb and Myclobutanil).

[iii]: from manufacturer's MSDS.

[iv]: as other plant growth regulator (Carbaryl).

[v]: as Triflumuron (benzoylurea).

[vi]: as Mancozeb (dithiocarbamate).

[vii]: as average of triazole fungicides (Triadimefon, Difenoconazole and Myclobutanil).

[viii]: as Flusilazole (triazole) + Syngenta MSDS, stating that it is a stable compound in soil.

Table III-47 shows the final fractions considered for each organic pesticide sprayed with an air-blast sprayer reaching the different environmental compartments in each IFP site under study. The fraction of pesticide reaching the surface water (f_w) is not shown in this table because it is always considered a fixed fraction (0.01%; see section III.2.4). For each active ingredient, the remaining fraction (adding up to 100%) either is degraded (by soil microorganisms or by sunlight) or remains on the plant.

Table III-47: Fractions of organic pesticides sprayed with and air-blast sprayer reaching air (f_a), groundwater (f_g) and soil (f_s) for each active ingredient and site under study, depending on timing and spray concentration.

Active Ingredient	*Site*	*Time* [a]	*Spray* [b]	f_a	f_g	f_s
Azinphosmethyl	IFP_HB_2	3	Dilute	65,6%	0,0%	0,5%
Bupirimate	IFP_CO_Avg	10	Dilute	66,9%	0,8%	1,5%
Bupirimate	IFP_HB_1	10	Dilute	67,4%	0,0%	3,7%
Bupirimate	IFP_HB_1	10	Concentrate	70,8%	0,0%	2,8%
Bupirimate	IFP_HB_2	10	Dilute	67,4%	0,0%	3,7%
Bupirimate	IFP_HB_2	10	Concentrate	70,8%	0,0%	2,8%
Bupirimate	IFP_HB_Avg	10	Dilute	67,4%	0,0%	3,7%
Buprofezin	IFP_CO_1	9	Dilute	97,8%	0,0%	0,0%
Captan	IFP_HB_1	1	Concentrate	4,5%	0,0%	0,0%
Captan	IFP_HB_2	1	Concentrate	4,5%	0,0%	0,0%
Captan	IFP_HB_Avg	12	Dilute	7,1%	0,0%	0,0%
Carbaryl	IFP_CO_1	11	Dilute	27,2%	23,8%	0,0%
Carbaryl	IFP_HB_1	11	Dilute	31,8%	0,0%	0,1%
Carbaryl	IFP_HB_2	11	Dilute	31,8%	0,0%	0,1%
Chlorpyrifos	IFP_CO_Avg	9	Dilute	64,8%	0,1%	0,0%
Chlorpyrifos	IFP_HB_1	9	Dilute	64,9%	0,0%	0,0%
Chlorpyrifos	IFP_HB_1	12	Dilute	46,6%	0,0%	0,0%
Chlorpyrifos	IFP_HB_2	9	Dilute	64,9%	0,0%	0,0%
Chlorpyrifos	IFP_HB_2	12	Dilute	46,6%	0,0%	0,0%
Chlorpyrifos	IFP_HB_Avg	9	Dilute	64,9%	0,0%	0,0%
Chlorpyrifos	IFP_HB_Avg	12	Dilute	46,6%	0,0%	0,0%
Cyprodinil	IFP_CO_1	9,5	Medium	63,9%	14,8%	1,0%
Cyprodinil	IFP_CO_Avg	10	Dilute	66,3%	0,3%	0,5%
Cyprodinil	IFP_HB_1	9,5	Concentrate	74,9%	0,0%	1,2%
Cyprodinil	IFP_HB_2	9,5	Concentrate	74,9%	0,0%	1,2%
Cyprodinil	IFP_HB_Avg	10	Dilute	66,5%	0,0%	1,9%
Diazinon	IFP_CO_Avg	12	Dilute	92,7%	0,0%	0,0%
Diazinon	IFP_HB_1	12	Dilute	92,7%	0,0%	0,0%
Diazinon	IFP_HB_2	12	Dilute	92,7%	0,0%	0,0%
Diazinon	IFP_HB_2	12	Concentrate	93,8%	0,0%	0,0%
Diazinon	IFP_HB_Avg	12	Dilute	92,7%	0,0%	0,0%
Difenoconazole	IFP_HB_1	11,75	Concentrate	64,1%	0,0%	0,0%
Difenoconazole	IFP_HB_2	11,75	Concentrate	64,1%	0,0%	0,0%
Difenoconazole	IFP_HB_Avg	12	Dilute	74,3%	0,3%	0,1%

Table III-47: Fractions of organic pesticides sprayed with and air-blast sprayer reaching air (f_a), groundwater (f_g) and soil (f_s) for each active ingredient and site under study, depending on timing and spray concentration. (continued)

Active Ingredient	***Site***	***Time*** [a]	***Spray*** [b]	f_a	f_g	f_s
Dithianon	IFP_HB_1	10	Dilute	20,8%	0,0%	0,0%
Dithianon	IFP_HB_1	10,5	Concentrate	27,9%	0,0%	0,0%
Dithianon	IFP_HB_1	11,75	Dilute	29,8%	0,0%	0,0%
Dithianon	IFP_HB_2	10	Dilute	20,8%	0,0%	0,0%
Dithianon	IFP_HB_2	10,5	Concentrate	27,9%	0,0%	0,0%
Dithianon	IFP_HB_2	11,75	Dilute	29,8%	0,0%	0,0%
Dithianon	IFP_HB_2	12	Concentrate	41,6%	0,0%	0,0%
Dithianon	IFP_HB_Avg	10	Dilute	20,8%	0,0%	0,0%
Dodine	IFP_CO_1	3	Medium	63,6%	7,3%	1,8%
Dodine	IFP_CO_1	9	Medium	51,7%	0,0%	0,1%
Dodine	IFP_CO_Avg	12	Dilute	58,8%	0,0%	0,1%
Dodine	IFP_HB_1	9	Concentrate	56,2%	0,0%	0,1%
Dodine	IFP_HB_1	9,5	Dilute	47,8%	0,0%	0,1%
Dodine	IFP_HB_1	12	Concentrate	72,8%	0,0%	0,1%
Dodine	IFP_HB_1	12	Dilute	58,8%	0,0%	0,4%
Dodine	IFP_HB_2	9	Concentrate	56,2%	0,0%	0,1%
Dodine	IFP_HB_2	9,5	Dilute	47,8%	0,0%	0,1%
Dodine	IFP_HB_2	12	Concentrate	72,8%	0,0%	0,1%
Dodine	IFP_HB_2	12	Dilute	58,8%	0,0%	0,4%
Dodine	IFP_HB_Avg	9,5	Dilute	47,8%	0,0%	0,1%
Dodine	IFP_HB_Avg	12	Dilute	58,8%	0,0%	0,4%
Flusilazole	IFP_HB_1	11	Concentrate	81,0%	0,1%	0,1%
Flusilazole	IFP_HB_2	11	Concentrate	81,0%	0,1%	0,1%
Flusilazole	IFP_HB_Avg	11	Dilute	85,9%	0,4%	0,1%
Kresoxim methyl	IFP_CO_1	10	Medium	23,9%	12,8%	0,0%
Kresoxim methyl	IFP_HB_2	10	Concentrate	27,9%	0,0%	0,0%
Lufenuron	IFP_CO_1	12	Medium	77,4%	0,0%	0,1%
Lufenuron	IFP_HB_1	11	Concentrate	64,1%	0,0%	0,1%
Lufenuron	IFP_HB_2	11	Concentrate	64,1%	0,0%	0,1%
Mancozeb	IFP_HB_1	11,75	Concentrate	5,8%	0,0%	0,0%
Mancozeb	IFP_HB_2	11,75	Concentrate	5,8%	0,0%	0,0%
Mancozeb	IFP_HB_Avg	12	Dilute	8,3%	0,0%	0,0%
Metiram	IFP_CO_1	10	Medium	14,8%	0,0%	0,1%
Metiram	IFP_CO_1	12	Medium	8,3%	0,0%	0,3%
Metiram	IFP_CO_Avg	12	Dilute	11,1%	0,0%	0,1%
Metiram	IFP_HB_2	10	Medium	14,8%	0,0%	0,1%
Metiram	IFP_HB_2	11	Concentrate	12,9%	0,0%	0,2%
Metiram	IFP_HB_2	12	Dilute	11,1%	0,0%	0,4%
Myclobutanil	IFP_CO_1	12	Medium	57,6%	0,3%	2,9%
Myclobutanil	IFP_HB_2	9,5	Concentrate	58,8%	0,2%	2,7%
Myclobutanil	IFP_HB_2	12	Dilute	58,3%	0,1%	4,1%
Naphtyl Acetic Acid	IFP_CO_1	11	Dilute	63,6%	15,8%	0,0%
Nitrothal isopropil	IFP_CO_1	11	Medium	27,7%	0,0%	0,0%
Nitrothal isopropil	IFP_CO_1	12	Medium	37,4%	0,0%	0,0%
Nitrothal isopropil	IFP_CO_Avg	12	Dilute	32,6%	0,0%	0,0%

Table III-47: Fractions of organic pesticides sprayed with and air-blast sprayer reaching air (f_a), groundwater (f_g) and soil (f_s) for each active ingredient and site under study, depending on timing and spray concentration. (continued)

Active Ingredient	***Site***	***Time*** [a]	***Spray*** [b]	f_a	f_g	f_s
Nitrothal isopropil	IFP_HB_2	12	Dilute	32,6%	0,0%	0,0%
Tebufenozide	IFP_CO_1	1	Medium	88,7%	0,0%	3,6%
Tebufenozide	IFP_CO_Avg	1	Dilute	84,0%	0,3%	3,2%
Tebufenozide	IFP_HB_1	1	Dilute	84,2%	0,0%	5,2%
Tebufenozide	IFP_HB_1	1	Concentrate	93,7%	0,0%	1,9%
Tebufenozide	IFP_HB_2	1	Dilute	84,2%	0,0%	5,2%
Tebufenozide	IFP_HB_2	1	Concentrate	93,7%	0,0%	1,9%
Tebufenozide	IFP_HB_Avg	1	Dilute	84,2%	0,0%	5,2%
Triadimefon	IFP_CO_1	1	Medium	17,8%	2,0%	0,1%
Triadimefon	IFP_CO_1	10	Medium	13,3%	29,0%	0,0%
Triadimefon	IFP_CO_Avg	11,75	Dilute	14,6%	20,0%	0,0%
Triadimefon	IFP_HB_1	11	Dilute	17,7%	0,1%	0,0%
Triadimefon	IFP_HB_1	11,75	Dilute	17,8%	0,1%	0,0%
Triadimefon	IFP_HB_1	11,75	Concentrate	18,5%	0,0%	0,0%
Triadimefon	IFP_HB_2	11	Dilute	17,7%	0,1%	0,0%
Triadimefon	IFP_HB_2	11,75	Dilute	17,8%	0,1%	0,0%
Triadimefon	IFP_HB_2	11,75	Concentrate	18,5%	0,0%	0,0%
Triadimefon	IFP_HB_Avg	11,75	Dilute	17,6%	0,7%	0,0%
Triflumuron	IFP_HB_1	12	Concentrate	96,5%	0,0%	1,4%
Triflumuron	IFP_HB_2	12	Concentrate	96,5%	0,0%	1,4%
Ziram	IFP_CO_1	1	Medium	11,1%	0,0%	1,6%
Ziram	IFP_HB_1	1	Dilute	15,1%	0,0%	2,3%
Ziram	IFP_HB_2	1	Dilute	15,1%	0,0%	2,3%
Ziram	IFP_HB_Avg	1	Dilute	15,1%	0,0%	2,3%

[a]: Month of application (number; weeks in .25)

[b]: Concentration (Dilute/Medium/Concentrate); see Table III-19 in page 82 for concentration ranges.

CHAPTER IV. SOIL QUALITY IN LAND USE IMPACTS. METHODOLOGY DEVELOPMENT FOR LIFE CYCLE IMPACT ASSESSMENT

"The surface of the earth crusted, a thin hard crust, and as the sky became pale, so the earth became pale, pink in the red country and white in the gray country.
In the water-cut gullies the earth dusted down in dry little streams" (p. 1)
"The owner men sat in the cars and explained. You know the land is poor. You've scrabbled at it long enough, God knows" (p. 33)
John Steinbeck (1939), The Grapes of Wrath

"Los indicadores territoriales […] deberán ser 'coherentemente subjetivos', y esto es, justamente, lo que los hace tan difíciles de establecer […]: todo el mundo ve cuando el agua rompe a hervir, pero pocos se atreven a decir en qué momento concreto deja de estar fría y pasa a estar caliente"[1]
Ramon Folch (1999), Diccionario de Socioecología*, p. 199*

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[1] Territorial indicators […] will have to be 'coherently subjective', and that is, precisely, what makes it so difficult to establish them […]: everybody can see when water starts boiling, but few dare to say what is the exact moment when it is not cold anymore and therefore starts being hot.

In the previous chapters the need to include local impacts in agricultural LCA has been highlighted. Land use related impacts were suggested as a likely main source of difference between different types of agricultural technologies (e.g.: conventional, integrated and organic), and therefore their development is a must if LCA is to be applied in the comparison of such types of agriculture. Land use impacts were described as affecting mainly biodiversity and life support functions, and many references for the biodiversity aspect have already been found in the literature. On the other hand, major research is still needed for the assessment of impacts on life support functions. The present chapter suggests an approach dealing with such impacts, and retakes the need of treating agricultural systems' analysis in a detailed and site-dependent way, as pointed out in chapter III.

In order to describe a new method to integrate life support functions into LCIA, a detailed understanding of how impacts are modelled within this framework is needed, and this is explained in the first section (IV.1). Impact assessment is explained using a bottom-up structure, from the inventory interventions to the areas of protection; this gives a deeper understanding on the many aspects affecting land degradation and how they are included in the impact analysis. In addition, a thorough description of the land functions (section IV.1.3) and how these have been included in the LCIA methodology so far (IV.1.4) are given in the first section. Then, section IV.2 explores deeper into the life support functions of land, and thus describes soil, soil quality and soil degradation. Also the methods used in other research fields for the assessment of soil quality are discussed in section IV.2.3.

As no consistent method for the inclusion of impacts on life support functions in LCIA is found in the literature, section IV.3 presents a new methodology for the assessment of life support functions in agricultural LCA, based on the soil organic matter (SOM) content[2]. The definition of this indicator is done following the structure suggested by SETAC, and presented earlier in the chapter (IV.1.2), in order to gain consistency with the rest of the approaches described in IV.1.4. Firstly, section IV.3.1 collects evidences from the literature showing the representation of the life support functions by SOM, with the aim of demonstrating the relevance of the indicator. Then, sections IV.3.2 and IV.3.3 discuss practical implications of the indicator, such as the ways of obtaining data for SOM

[2] The proposal of an indicator for life support functions of land based on SOM has been presented by the author in an international forum; see Milà i Canals *et al*. (2000, 2001). Besides, the author has actively participated in the SETAC Working Group on LCIA, in the taskforce on resources and land use, where he could have interesting discussions on this issue with other experts in LCIA. The results of this working group are about to be published in Udo de Haes *et al*. (2002).

levels and evolution, as well as the operations and calculations needed for the application of the indicator in LCIA. IV.3.4 reviews the consistency of the indicator in the light of the framework defined in IV.1.2, and IV.3.5 presents an evaluation of the SOM indicator based on ISO requirements and other relevant literature. Practical allocation rules when describing impacts from land use are given in section IV.3.6 and a brief consideration on the implications of the SOM indicator for the global warming impact category is done in section IV.3.7. The practical implications for the application of the SOM indicator for LCIA are dealt with in section IV.4. This section describes the use of SOM models in the implementation of the indicator (IV.4.1), and further illustrates the indicator with a theoretical example of application (section IV.4.2). Conclusions for the chapter are given in section IV.5.

IV.1. Framework for Life Cycle Impact Assessment of land use related impacts

The needs for Life Cycle Impact Assessment (LCIA) of land use impacts are discussed in this section. First, a general overview of LCIA is given, and then a detailed description of how land use impacts can be assessed for LCIA is included. Finally, a brief review of the state of the art of methods to include land use impacts in LCIA is provided.

IV.1.1. Life Cycle Impact Assessment

Life Cycle Impact Assessment (LCIA) is the third phase of LCA, and the one previous to the final interpretation of the results. The main objective of this phase is to convert the long list of *environmental interventions*[3] resulting from the life cycle inventory into a reduced list of effects on elements that are relevant to society. As represented in Figure IV-1, this is done through the description of the *environmental mechanism* (or impact chain), defined by the ISO 14042 as the total of environmental processes which link the environmental interventions to the *endpoints* of a type of impact (ISO 2000). The endpoints are thus the physical elements of an environmental mechanism that are in themselves of value to society (Udo de Haes & Lindeijer 2002). Examples of endpoints are forests or particular species (due to its value –be it functional or intrinsic- to society), fossil fuels (because we use them, and in case they are depleted we would need to find substitutes), etc. Also aspects of human health such as lifetime are endpoints that society wants to preserve.

Endpoints or Areas of Protection

These endpoints can be grouped in *Areas of Protection*. Generally, three of such areas have been distinguished: Human Health, Natural Environment (or “ecological health” or “ecosystem quality”), and Natural Resources (Fava *et al*. 1993), and some impact assessment methods follow this structure (e.g.: Goedkoop & Spriensma *et al*. 1999). Many other classifications of areas of protection have been suggested so far, basing these classifications on physical characteristics (damage to atmosphere, hydrosphere… or biotic resources, abiotic resources…) or societal values (man-made environment, natural environment… functional or intrinsic values…). I will keep to the classification recommended by the SETAC working group on LCIA (Udo de Haes & Lindeijer 2002), which is based in societal values and includes a detailed description of the area of protection “Natural Environment” (see Figure IV-2).

[3] The environmental interventions are the physical elements crossing the border between the product system and the environment, such as extracted natural resources entering the product system and hazardous emissions leaving it and entering the environment (Udo de Haes *et al*., 1999; Udo de Haes & Lindeijer 2002). ISO 14042 uses the term *elementary flows* in this context. Also *environmental exchanges* (Wenzel *et al*. 1997) and *stressor* (Fava *et al*. 1993) have been suggested. I will keep to the terminology *environmental intervention* as is suggested by the SETAC Working Group on LCIA (Udo de Haes *et al*. 1999).

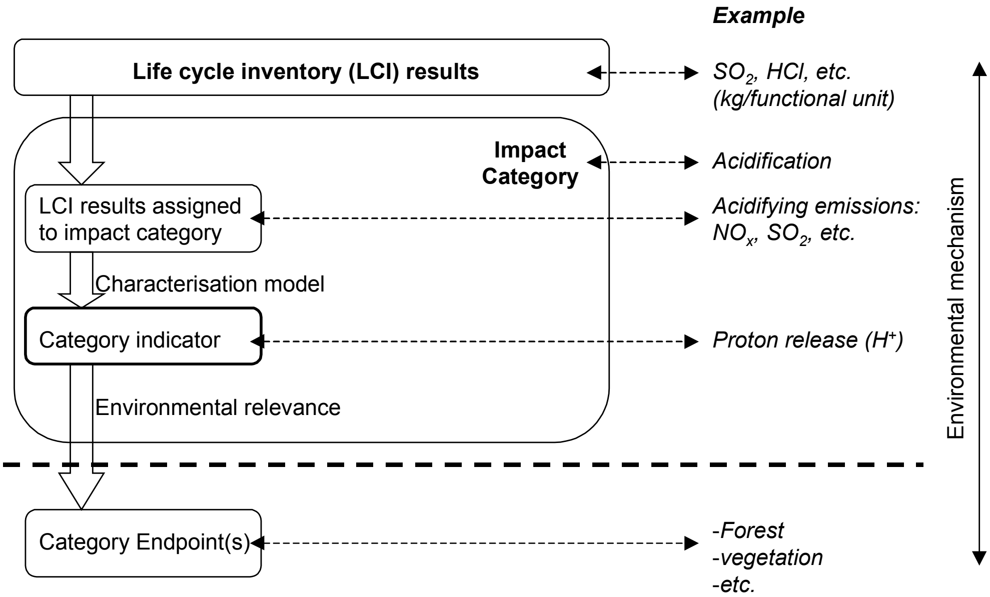

Figure IV-1: Concept of category indicators and environmental mechanism (ISO 2000).

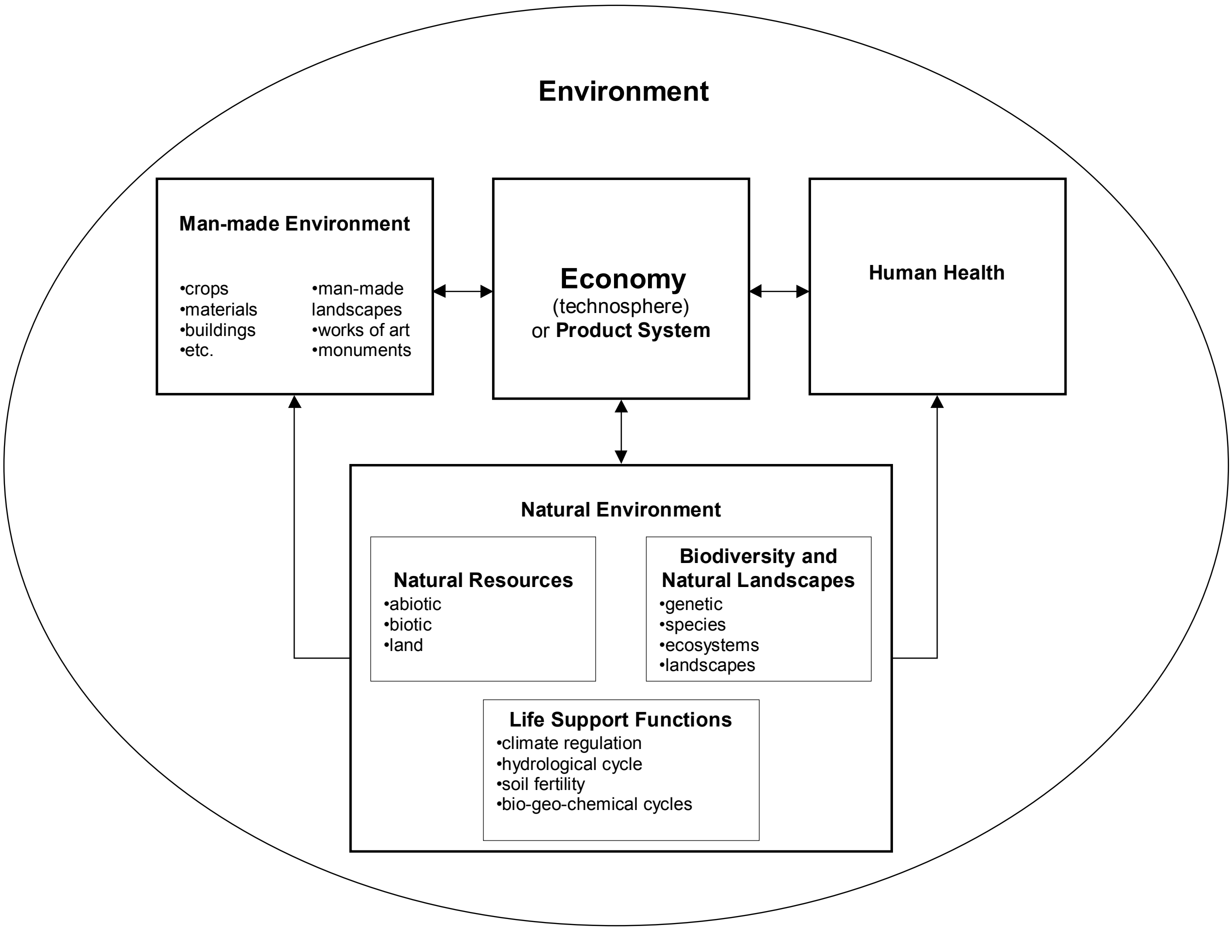

Figure IV-2: Classification of Areas of Protection according to societal values (Udo de Haes & Lindeijer 2002).

The description of the environmental mechanism leads to the effects of the environmental interventions on the areas of protection. The user of the information provided by the LCIA (e.g.: the decision maker) usually includes a further step, aimed at determining the relative importance of each impact category or area of protection. This is done because different alternatives tend to imply trade-offs between impact categories. In LCA, this step is commonly known as valuation or weighting, because it implies "aggregating indicator results across impact categories using numerical factors based on value choices" (ISO 2000).

In Figure IV-2, the Natural Environment has been split up using societal values into three areas of protection. Biodiversity and Natural Landscapes covers most of the intrinsic values of the Natural Environment, while Natural Resources and Life Support Functions have predominantly a functional value to society (Udo de Haes & Lindeijer 2002). The latter represent the dynamic nature value of an area (Lindeijer 2000b). In this framework, then, the Natural Environment is supporting the Man-made environment (biotic productivity for crops, source of materials, basement for buildings, etc.) and Human Health (regulating climate, providing fresh and clean water, etc.). The relation with the product system and more generally with the global Economy is obviously bi-directional, as the Natural Environment provides resources for the product system (both physical and functional) and receives its emissions. It is often suggested that impact assessment should be focused on the intrinsic values of endpoints, while functional values should be included in the definition of the functional unit or the performance of the analysed system. Nevertheless, it is also recognised that as long as these functional values are not included in the definition of the system's function, damage to them must be assessed using an impact assessment procedure. An example would be the definition of a functional value as "production of 1 tonne of export and local apples while maintaining the productive value of soil". As the inclusion and evaluation of functional values in a functional unit is very difficult, the present chapter suggests a method for their assessment as an impact category.

The concept of indicators

Environmental indicators are instruments for communication on the ecological state or change of a region, an enterprise's behaviour, etc. Thus, they need to be simple constructions from available environmental statistics, which means that they must simplify a complex reality (Smeets & Weterings 1999). From a practical point of view, they are measurable surrogates of environmental attributes that cannot be measured directly. In the field of LCIA, ISO 14042 (ISO 2000) defines a category indicator as a quantifiable representation of the impact category.

As indicators are not real measures of the current condition that wants to be assessed, when working with indicators a special awareness of the conditions in which measurements and modelling were made is needed. Also the scale at which answers are needed has to be known. Hoosbeek & Bouma (1998) suggest a framework for the assessment of the indicators that are used for soil and land quality assessment, based on a 3-axial diagram addressing the scale, complexity and transferability of the indicator (see Figure IV-3).

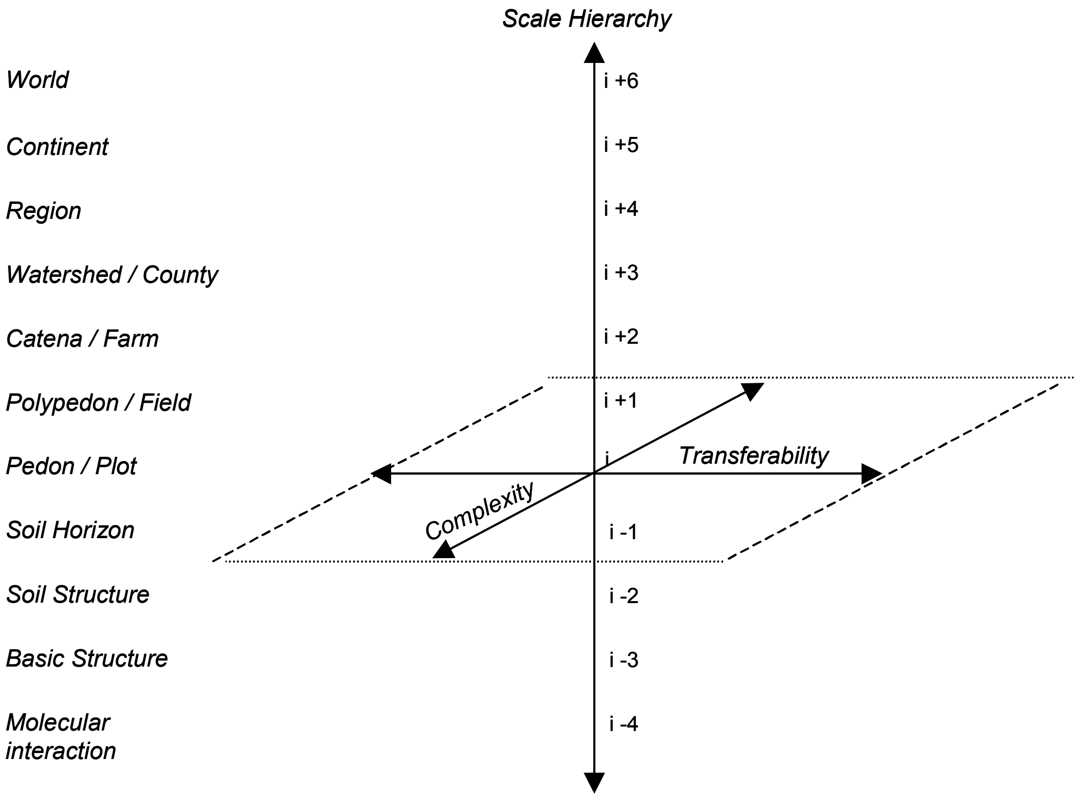

Figure IV-3: Classification of soil and land quality indicators based on scale level, complexity and transferability (Hoosbeek & Bouma 1998).

Ideally, an indicator to be used in LCIA should comply with the characteristics presented in Table IV-1.

Table IV-1: Characteristics that an indicator should have.

Concept	*Decription*
Representation	The indicator should be representative of the endpoints that want to be preserved
Measurability	As any indicator, it should be a measurable surrogate of the attributes defined as relevant for the area of protection
Consistency	It should be consistent with the framework for LCIA and the objectives and scope of the study. Furthermore, the indicator should be sensitive to the impacts inflicted on the endpoint
Applicability	For the sake of simplifying the implementation of LCIA, it should be easily applicable by non-experts, and data should be readily available that make its application possible
Site-dependency	The indicator should be site-dependent when the impact category requires so and it is relevant for the goals of the study
Scale	Indicators have to work at the relevant scale where answers are needed (see Figure IV-3)
Transferability	In a globally applied tool as LCA, the indicators used should be transferable and internationally accepted in order to allow for comparisons between different LCA studies (see Figure IV-3). They should also allow for aggregation over impacts in different situations (e.g.: in different ecosystems, when ecosystem degradation is being assessed)

The three last characteristics are addressed by what is known as Generic Application Dependency (Udo de Haes *et al*. 2002). This concept refers to the fact that the indicator should be able to work as a site-dependent one when the application requires so, but it should be also possible to apply it in a site-generic way when needed. For example, toxicity indicators are usually derived for average conditions for the exposure routes, although site-dependent information might be suited in them in order to allow for site-dependent requirements.

In summary, when an impact category has to be described the endpoint(s) affected by this category must be determined, along with the indicators suitable for this endpoint(s), the environmental interventions that affect them, and the environmental mechanism through which they operate (ISO 2000).

IV.1.2. Land use impacts in LCIA

In order to present the methodology needed for the assessment of the impacts derived from land use, some terminology and a framework to characterise these impacts are required.
A specific "land use" is generally defined as a human activity that occupies land area. According to the Food and Agriculture Organisation (FAO 1976), a type of land can be defined as:

> *"An area of the earth's surface, the characteristics of which embrace all reasonably stable, or predictably cyclic, attributes of the biosphere vertically above and below this area including those of the atmosphere, the soil and underlying geology, the hydrology, the plant and animal populations, and the results of past and present human activities, to the extent that these attributes exert a significant influence on present and future uses of the land by man"*

The activities that humans perform on land are thus affected by the qualities (resources) of that certain piece of land. These *land attributes* are diverse and allow for the maintenance of many different activities or *land uses*, which are associated to what is known as *land functions*. In LCA terminology, a land use would be a specific type of environmental intervention defined in the LCI results, which will change the ability of land to perform its functions (Audsley *et al*. 1997). This change in performance of land's functions is called "land use impact"[4] in the field of LCA. Even though the first LCIA methodologies mentioned land use as an area impacted by product systems, they only considered the amount of land used (ha) as an indication of the impacts. Today it is acknowledged that apart from the amount of land being used (occupation), it is this change in quality that has to be assessed in LCA. Otherwise, it would be like adding up all the chemicals emitted to air and assuming that all of them have the same effects, without taking into account their different environmental relevance.

[4] In some occasions, also the term "land use" has been used to denote the impacts derived from land use in the field of LCA (Udo de Haes *et al*. 1999), which may lead to confusion.

Occupation and transformation

According to the framework defined by the SETAC Working Group on LCIA (Lindeijer *et al*. 2002), land use impacts affect land through physical occupation and/or transformation of its quality. Land quality is generally measured by the performance of its functions, as is further developed in next section (IV.1.3. Land quality: assessing the functions of land). Figure IV-4 shows the practical implications of these aspects for land use impacts:

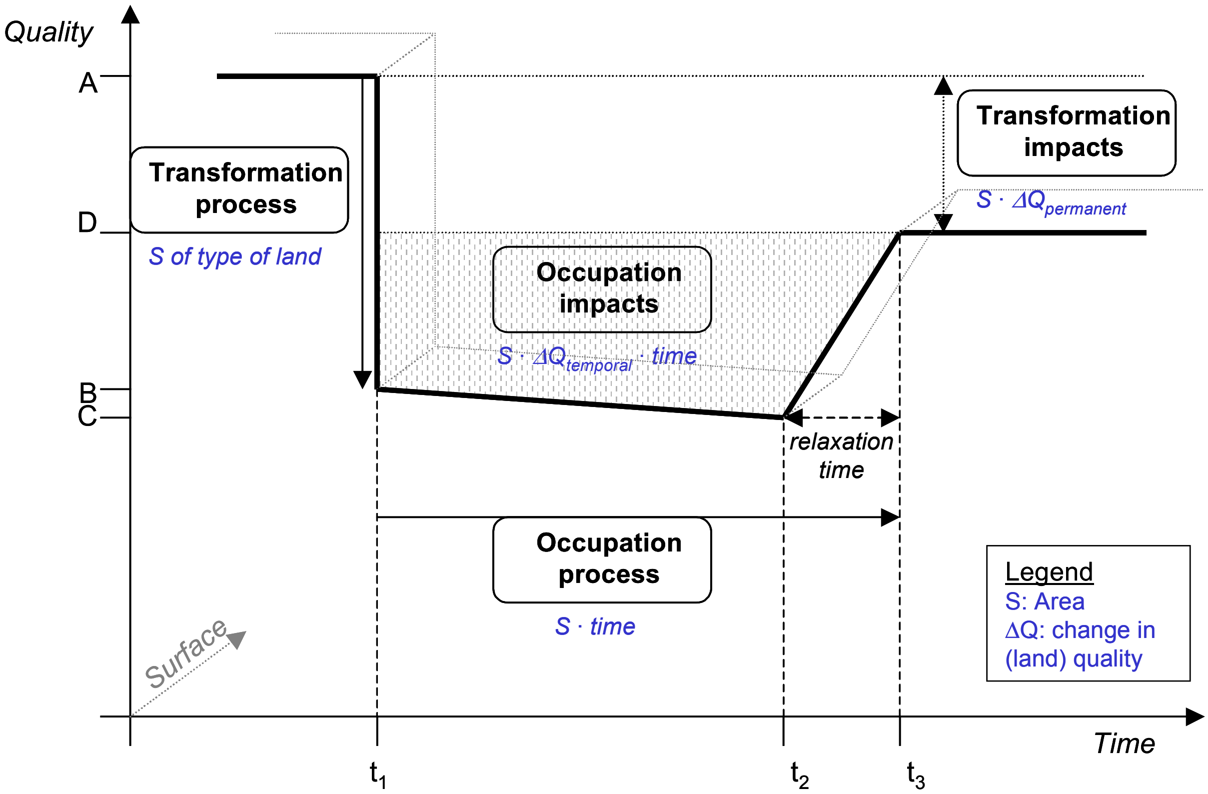

Figure IV-4: The different aspects of land use impacts (adapted from Lindeijer *et al*. 2002).

In Figure IV-4, land quality is expressed in the vertical axis, and time in the horizontal axis. Surface may be expressed as the third axis, which will not be depicted in the following pictures for the sake of simplicity. The black bold line represents the evolution of land quality with time, and several moments are depicted in this evolution: t_1 is the beginning of a land use; t_2 the end of the same land use, and t_3 is when a natural (or human-induced) recovery process (relaxation) reaches a steady state. The *transformation process* is thus the change of a land area into a new type of land, which meets the characteristics required by the new land use, and can be measured by the surface of transformed land (ha). The new land use is performed during the *occupation process*, measured by the area used during a length of time (ha · years). The occupation process does not only take into account the actual occupation, but also the *relaxation time* (from t_2 to t_3), i.e.: the amount of time it takes land to recover to a steady state (which can be the original –"A"- or a new one as level "D" in Figure IV-4). The steady state that can be reached after a degradation of the original state is known as *relaxation potential*. The process of recovery to the relaxation potential (from C to D) is known as re-naturalisation (or relaxation), and it is part of the transformation process. If the re-naturalisation process is not complete (i.e.: the original steady state "A" is not reached, the impact not reversed), a permanent change in quality known as a (permanent) *transformation impact* is produced. This transformation impact is measured using the amount of surface that has changed and the dimension of the (permanent) change ($\Delta Q \cdot$ ha), i.e.: the difference between the original state and the relaxation potential. In other cases a full recovery is observed ("D" equals "A"), i.e.: the impact is reversible. Then, no transformation impact is considered. Müller-Wenk (1998) talks about "transforming" type of land use when transformation impacts do occur, and about "maintaining" type of land use when no transformation impacts are observed. Different types of impacts according to their reversibility are further explained in Table IV-2, which is based on the descriptions from The Dobris Assessment (Teller *et al*. 1995). Different degrees of impact can thus be identified, depending on the ecological responses found in each case, i.e.: the degree of reversibility of the degradation effects.

Table IV-2: Types of land degradation according to the reversibility of the effects.

Nature of the effect	***Description (Teller* et al. *1995)***	***Type of land use (based on Müller-Wenk 1998)***
Naturally reversible	When the threat is removed land quality returns to initial state. In this case, the question is the time scale required (how long does it take to reverse the effects?).	Maintaining
Reversible with proper management	Land quality can still be fully recovered, but human action is required.	Maintaining
Convertible to another desirable state	Proper human management may lead to a steady state of lower quality than the initial, but still acceptable for other uses.	Transforming
Irreversible	Any of the soil (land) functions is permanently impaired.	Transforming

Finally, *occupation impacts* are those physical changes originating from the occupation process. Generally speaking, they can be considered as the prevention of re-naturalisation (i.e.: preventing land from reaching the potential state), because the fact that land has a lower quality due to the land use under study is an impact in itself. Occupation impacts are measured by the amount of area affected, the difference in quality that is maintained through the occupation process (ΔQ)[5] and the duration of the occupation process (ha $\cdot \Delta Q \cdot$ years).

The discussion on occupation / transformation impacts presented above and illustrated in Figure IV-4 can be related to the more general definition of impacts related to the *use* or *consumption* of resources. Heijungs *et al*. (1997) clarify this issue for resources with the following figure based on Finnveden *et al*. (1996):

[5] This difference in quality may vary over the occupation process, such as in Figure IV-4: from B to C and then to D.

	Use	Consumption
Depletion	-	Deposits, funds
Competition	Flows, funds	-

Figure IV-5: Relationship between use/consumption, depletion/competition, and deposits/funds/flows (Heijungs *et al*. 1997).

Land is a resource that is being used, and so it cannot be depleted. Generally one can only talk about competition for the resource land. But when land quality is altered (generally degraded), then that piece of land with a certain quality is no longer available, i.e.: it has been depleted (at least in the scope of the study, until quality has been recovered). Thus, even though *land* is a flow resource, *land quality* may be considered a fund susceptible to be depleted (when the annual degradation –consumption- exceeds annual regeneration, see Finnveden *et al*. 1996) or temporally impaired.

The Reference State – Relaxation Potential

As presented in Figure IV-4, occupation impacts are measured as the difference between the present state and the relaxation potential (times the total occupation time times the area). Thus, the relaxation potential is being chosen as the reference state to which impacts are related. Nevertheless, it is arguable whether the relaxation potential is the correct reference state; some authors have suggested that the state prior to land use should be the reference (Baitz *et al*. 1999), while others say that the "natural" state should be the reference (e.g.: in the *Hemeroby* concept, see Brentrup *et al*. 2002 for a review of the application of this concept in LCIA). Weidema & Lindeijer (2001, p. 15) give an interesting discussion on why the Reference State should be put in the relaxation potential. In short, it can be said that it is a practical way of distinguishing between permanent changes in quality (transformation impacts) and temporal changes (which are relevant as occupation impacts, but not from a transformation point of view). Therefore, the current relaxation potential is chosen as a consistent way of defining the reference state in the framework for land use impact assessment in LCA.

Relaxation time

On the other hand, the temporal scope of the study will define the time that is considered for relaxation. If the impacts were analysed to the infinite, then there would probably always be a full recovery, and no transformation impacts would be considered. Oppositely, not considering the relaxation time at all would imply a zero recovery; in this case the occupation process would be underestimated and the transformation impacts overestimated.

Udo de Haes *et al*. (1999) suggest that characterisation factors for LCIA should be calculated for infinite time without discounting impacts happening in the future. In the case of land use impacts, this would probably imply not considering transformation impacts, as discussed above. They also recommend checking whether a short period such as 100 years would yield considerably different results, and if this is the case characterisation factors for a 100 year period should also be calculated. Land use impacts should thus be modelled allowing for a recovery time of at least 100 years to see whether there are transformation impacts.

IV.1.3. Land quality: assessing the functions of land

The vertical axis in Figure IV-4 can be measured with suitable indicators of land's functions because a land use impact is a change in the performance of such functions. Many attempts have been made to list all the relevant land/soil functions. The Dobris Assessment (Teller *et al*. 1995; see also Blum 1990) divides these into ecological functions and socio-economic functions (see Table IV-3).

Table IV-3: Ecological and socio-economic functions of land.

Type of functions	***Teller et al. 1995***	***Lindeijer et al. (2002)[a]***
Ecological	▪ Production of biomass ▪ Filtering, buffering and transforming ▪ Gene reserve and protection of flora and fauna	▪ Habitat for non-human life, i.e.: the ability to sustain biodiversity ▪ Base for food and other biotic production ▪ Element in the freshwater circuit, including the influence on the water flows between the different media of the water circuit (infiltration, surface runoff…) ▪ Element in the global energy circuit, including the effects on land albedo and the carbon cycle ▪ Microbiological transformations
Socio-economic	▪ Support to human settlements (housing, infrastructure, recreation, and waste disposal) ▪ Source of raw materials, including water ▪ Protection and preservation of cultural heritage	▪ Habitat for humans (including land stability) ▪ Base for food and other biotic production ▪ Place of abiotic resources deposits ▪ Sink for wastes and pollutants ▪ Other functions: aesthetic welfare and cultural values

[a]: Classification into ecological / socio-economic functions is done by the author.

As it can be seen in Table IV-3, the listing in Lindeijer *et al*. (2002) includes all of the functions mentioned in the Dobris Assessment, but stresses others closely linked to the role of soil in closing substance cycles. Note also that the production of biomass has been repeated in the listing by Lindeijer *et al*. (2002), in order to stress that this is both an ecological function and has a deep socio-economic impact through agriculture.

Areas of Protection affected by land use impacts

According to the functions defined above, the second working group on Life Cycle Impact Assessment in SETAC (Udo de Haes *et al*. 1999) has defined three main subcategories to distinguish between the different endpoints affected by land-use related impacts:

- Subcategory 1: increase of land competition
- Subcategory 2: degradation of life support functions
- Subcategory 3: bio-diversity degradation (due to land use)

Hence, the land use impacts can generally be related to the following Areas of Protection (see Figure IV-2):

- Natural Environment: Here, impacts on the three sub-areas of protection can be distinguished:
  - Natural Resources: land as a spatial resource is the issue here. Two types of impact could occur; first, there might be depletion of land (quality) if it is irreversibly altered or lost (e.g.: through severe erosion) producing a transformation impact. Also, competition for the resource “land” between different uses and users (including nature) occurs when land is occupied (occupation process). this competition is not considered as an environmental impact because its effects are mainly of economic relevance (Lindeijer *et al*. 2002).
  - Biodiversity and Natural Landscapes: land occupation and/or transformation may lead to changes in biodiversity and landscapes (e.g.: changing species composition of an ecosystem, reducing a habitat size, etc.).
  - Life Support Functions: land occupation and/or transformation may change the ability of land to perform these functions, either temporarily (occupation impacts) or irreversibly (transformation impacts). This of course may eventually affect Human Health (e.g.: through decreased availability of food)
- Man-made environment: the degradation of man-made landscapes as part of the cultural heritage and as a result of land use is considered here. These impacts have not received much attention in LCIA until now. In addition, the effects on crops from impacts on life support functions are relevant here.
- Human Health: land uses may change the contribution of land to different impacts that affect Human Health. E.g.: increased use of fertilisers by agriculture leads to emissions of nitrous oxide (N_2O), which contributes to Global Warming; the possible effects of Global Warming to human health have been long recognised (through increase in the incidence of tropical diseases, etc.). These impacts, though, are considered under other *ad hoc* impact categories (e.g.: Global Warming, Human Toxicity…), and will not receive further attention in this dissertation.

IV.1.4. State of the Art in LCIA of land use impacts

Lindeijer (2000a) gives an extensive review of land use impact assessment methods in LCIA, and Heijungs *et al*. (1997) present an earlier one. Therefore, no attempt of completeness is done here. Rather, a mention of the main methods will be done focusing on the ones that assess impacts on the life support functions of land. This review covers the following items:

- **Representation**: Areas of Protection / Land functions that are covered by the method.
- **Mechanism & Model**: which are the parameters and indicators used in the environmental mechanism?
- **Choices & Assumptions**: hypothesis made in the approach and consequences for the validity of the method.
- **Applicability**: mainly related to data availability.
- **Conclusions**.

Heijungs *et al*. 1992 (CML 1992) (discussions in Heijungs *et al*. 1997)

Representation	Occupation and transformation
Mechanism & Model	Occupation: ($m^2 \cdot yr$) Transformation: (using a classification of qualities of land use: I. natural systems, II. modified systems, III. cultivated systems, IV. built systems and V. degraded systems); consider only changes due to the activity under consideration. Dimensionless changes (from class III to IV, characterisation factor of 1)
Choices & Assumptions	Categories are very general, and classification in one or the other may seem ambiguous. Changes from the first three types to the last two are regarded as damage, and aggregated without weighting.
Applicability	Direct, although the practitioner may introduce important biases when classifying land uses.
Conclusions	Does not directly consider occupation impacts and effects on life support functions. It may be enough as a first rough approximation, although it is not very sensitive for the comparison of similar systems, and does not allow for the assessment of maintaining types of land uses.

Fava *et al*. 1993 (SETAC's 'Code of Practice')

Representation	Only the competition aspect is considered (land as a spatial resource) and is treated as a depletion impact; considers the inclusion of competition impacts only in the cases where "a clear depletion [sic] problem exists for a specific usage", and mentions the case of landfills.
Mechanism & Model	A direct link between m^2 used and land competition is assumed. The time dimension of land occupation ($m^2 \cdot y$) is not acknowledged. Other indicators such as m^3 for landfill space are possible when relevant.
Choices & Assumptions	All areas used cause the same competition; mainly landfill occupation is in mind. The types of land that are distinguished may be arbitrary and affect the results to a great extent.
Applicability	There is a lack of consistency with the framework discussed above (only m^2, not $m^2 \cdot$year). The main problem is that they consider land as a stock resource, when generally it should be considered either a flow (when occupation is the issue) or a fund (when land quality is being depleted); see Figure IV-5.
Conclusions	Depletion of land area is not consistent with the framework presented above, and so this method is not applicable. Nevertheless, the formula suggested may be useful in the case of the amount of soil that is lost (actually, this is applied in Cowell 1998, see below).

Cowell 1998

Representation	The methods presented in the PhD dissertation mainly cover the aspects related to the life support functions of land, biodiversity, and soil as a scarce resource (natural resources)
Mechanism & Model	Even though Cowell cites many different parameters related to crop productivity (nutrients, weeds, pathogens, salinity, pH) she suggests that these are usually maintained in sustainable ranges in order to ensure production. The operations done for maintenance already account for these aspects, and thus they do not need to be included in LCIA unless there is a reason to believe they are problematic in a special study. Two indicators are suggested to include the life support functions in the analysis: Organic Matter Indicator (inverse of any addition of OM to soil) and Soil Compaction Indicator (Area · Weight of vehicles working in the area · Time of work in hours/ha). Soil composition effects on compaction are not included. The recovery time is not included. No relation to a reference state; all OM addition is positive. In order to assess impacts on Biodiversity, a Physical Habitat Factor is defined for each ecosystem. This includes aspects related to the area of the ecosystem, rare species, total number of species, and the Net Primary Productivity. The PHF is complemented with a Physical Management Factor in order to calculate Physical Habitat Degradation. The resource aspect is considered by assessing the amount of soil that is eroded from the system (either by direct measurement, calculation with well-established models[6], or estimation). Then, the static reserve life of soil is calculated as 104 (94-114) years, in order to characterise soil loss with other abiotic resources. Further reference to these methods may be found in Cowell & Clift (2000).
Choices & Assumptions	Life support functions are well represented by organic matter additions and soil compaction. Any addition of organic matter is positive for the maintenance or improving of soil quality. Includes productivity as an aspect of Biodiversity, which according to Udo de Haes & Lindeijer (2002) should mainly describe the inherent values of land (productivity represents a functional value). Trade-offs between ecosystems are accepted.
Applicability	Even though models need to be applied for most of the LCA applications in the resources and life support functions indicators, data will generally be available for doing so, at least in the scope of agricultural LCA. In the case of biodiversity, data availability will be a problem in most countries; nevertheless, Cowell applies the method to the UK and suggests the data sources necessary for doing so in other countries.
Conclusions	Good inclusion of life support functions, although the occupation impacts are underestimated because the relaxation time is not considered. No procedure for weighting the organic matter indicator and the soil compaction indicator is suggested. Good to compare maintaining land uses (e.g.: different agricultural systems).

[6] The Revised Universal Soil Loss Equation (RUSLE) is mentioned.

Mattsson *et al*. 1998 (further referenced in Mattsson *et al*. 2000)

Representation	The functional values of land are well represented with a series of indicators related to the life support functions. Biodiversity and aesthetic landscape value are treated as well, though in a more qualitative way.
Mechanism & Model	Indicators for the life support functions (mainly focused at productivity) are suggested and quantified where possible: erosion, hydrology, soil organic matter, soil structure, nutrient balance, soil pH, and heavy metals. Some of these are found to be not applicable due to lack of data, while data sources are suggested for the most relevant ones (erosion, SOM, structure, pH, and nutrient –P & K- status). Nevertheless, the information is used separately, and no attempt of aggregation of the indicators is done. For biodiversity, the number of species is quantified before land transformation and for current land use, but no calculation method is suggested. The aesthetic value of landscape should be qualitatively assessed using regional references.
Choices & Assumptions	The different productivity indicators are to be used independently, without any weighting. Higher number of species per area is of higher value, and thus changes in areas with higher biodiversity (e.g.: tropical forest) are worse. The reference state is thus set at the state prior the activity.
Applicability	The applicability of productivity indicators depends heavily on local data availability, although models do exist for erosion, which is one of the most relevant indicators. For biodiversity, the potential vegetation type needs to be known, and local data needs to be available.
Conclusions	They insist on the need of basing land use impact assessment on a mix of qualitative and quantitative indicators, which will be possible in site-specific conditions. Nevertheless, the obtaining of all the data required is difficult, and the calculations that need to be performed with the different indicators are not explained. Some of the productivity indicators might be superfluous, as depend heavily on each other (e.g.: structure and to some extent pH are strongly correlated to SOM).

Müller-Wenk 1998

Representation	The method is focused on the impacts on “ecosystems” through the indicator threatened vascular plant species (the intrinsic value of biodiversity, not the life support function of it). Life support functions of land and the resource value of land are left out of it.
Mechanism & Model	The impacts are fully quantified via a damage function that links “high intensity” land use with loss of species. There is no regionalisation. A direct link between occupation and regional species diversity loss is obtained by estimating 2 points of the damage function. Average relaxation time is suggested for transforming type of land uses, allowing for extended occupation impacts on biodiversity.
Choices & Assumptions	Assumes that habitat integrity is a good indicator, and chooses not to combine it with loss of productivity or loss of soil as a resource. These assumptions are justified in the scope of Central Europe. Based on 2 datasets of endangered species (Switzerland, Germany). Assuming full recovery after transformations thus allowing adding up occupation and transformation damages.
Applicability	Simple due the classification of land use in 4 basic types. Data usually exist, though the method is developed for Switzerland and Germany only. Some of the assumptions might not be so valid for other countries, even in Europe.
Conclusions	The method is promising for the impacts of land use on biodiversity, even though some assumptions should be revised in the scope of non-Central European countries. In agricultural LCA this method is not enough because impacts on the life support functions of land are highly relevant. Also the resource aspect of land should be included somehow.

Lindeijer *et al*. 1998 (further referenced in Blonk *et al*. 1997 and Lindeijer 2000b)

Representation	Biodiversity (vascular plant species) and LSF (fNPP assumed to be an indicator for C-cycling and soil quality)
Mechanism & Model	Biodiversity is measured with the vascular plant species (α, number of plant species per m^2). Full quantification via $A*t*(\alpha_{ref}-\alpha_{act})/\ \alpha_{ref}$ for occupation, and $A*(\alpha_{ref}-\alpha_{act})/\ \alpha_{ref}$ for transformation. A rough regionalisation is possible, and they provide estimates of α values. Life support functions are measured with the (free) Net Primary Productivity (fNPP). Full quantification via $A*t*(fNPP_{ref} - fNPP_{act})$ for occupation, and $A*(NPP_{ini} - fNPP_{fin})$ for transformation. A rough regionalisation is possible, and they provide estimates of fNPP values. By subtracting the harvested C, the focus is on the natural processes.
Choices & Assumptions	Assuming that vascular plant species are a good indicator of biodiversity. Assuming a linear relationship between fNPP difference and life support functions. Impacts of occupation and transformation are left separate by not assuming full recovery, but no aggregation method is given for occupation and transformation impacts, resulting in four scores results.
Applicability	Good for land use types in mid-Europe, moderate for outside Europe. The authors provide rough data (both α and fNPP values of different biotopes) for a first screening application. More detailed fNPP data is hard to get.
Conclusions	This represents possibly the most advanced method of those available in the literature, both in the refinement of the method, the consistency with the LCIA methodology, and the provision of data for its application. Assessing how much biomass is left for nature's functioning and life support functions (i.e.: fNPP) may be used to compare maintaining land uses. Nevertheless, Burger & Kelting (1999) suggest that growth-based indicators (such as NPP and thus fNPP) might not be a good approach to assess soil quality, because they are affected by many other factors (weather, fertiliser use, etc.). Bouma (2002) also addressed this issue, and suggests that low-quality soils may have a high productivity due to excellent management and vice versa. Indeed, yield should be seen as an indicator of the short-time effects of management practices, but is not a proper indicator of the long-term system's sustainability. Section IV.2 concludes that life support functions of land depend on soil, and therefore soil-based indicators should be preferable for these functions.

Baitz *et al*. 1999

Representation	Focused on functions of interest to humans. Other functions, such as protection of biodiversity (included with the *Hemerobiestufen*[7] approach to account for potential stability) are less dominant. Intrinsic values of land (landscape…) are intentionally left out to avoid value judgements. Full quantification of 11 different life support functions, fully regionalised, comprehensive.
Mechanism & Model	Full quantification via $A*t*(Q_{ref} – Q_{act})$ for occupation impacts. A 'fulfilment' mark is given to each of the functions considered (erosion resistance, filter and buffer function, microbiotic transformation, groundwater function, rainwater drain, biotic output, immission protection, potential stability), and then they are all equally weighted to get the value for land quality at each moment (Q_{act}). They suggest that different weights may be allocated according to regional preferences. No quantification of the relationship with endpoints is given. The recovery time is included in the calculation.
Choices & Assumptions	Equal weights for land functions are acceptable. Net transformations are neglected (full recovery is assumed).
Applicability	Depends on local data availability. The authors provide data for a general classification of indicator values.
Conclusions	The detailed functions description and the considering of a full recovery make it suitable for comparing maintaining land uses that cause reversible degradation, such as most agricultural systems. Nevertheless, equal weighting of soil functions is not always acceptable, and some of the functions are actually interdependent, which suggests some double counting might happen.

Goedkoop & Spriensma *et al*. 1999 (EcoIndicator 99)[8]

Representation	It is suggested to describe Ecosystem Quality in terms of energy, matter and information flows. In the Eco-indicator 99 (EI99, from now on), they concentrate on the level of disruption of the *information flow* in order to assess Ecosystem Quality, i.e.: on impacts on biodiversity, at species level. Hence, neither account of life support functions nor the nature of land as a spatial resource is done.
Mechanism & Model	Full quantification based on 5 land use types using a modification of the formula's of Köllner (2000), no regional differentiation. The Potentially Disappeared Fraction (PDF) of species (vascular plants) is used as a parameter to represent the effects of land use (as well as acidification and eutrophication) on vascular plant populations in an area. PDF can be interpreted as the fraction of species that has a high probability of no occurrence (disappearance) in a region due to unfavourable conditions.
Choices & Assumptions	It is assumed that the diversity of species is an adequate representative for the quality of ecosystems. Assume that the transformations will be recovered, and then add the recovery time to the occupation time.
Applicability	Data available for Europe.
Conclusions	Widely applied currently, because it is a default method in SimaPro[9]. Nevertheless, no inclusion of impacts on life support functions makes it useless for comparing e.g.: agricultural technologies.

[7] "Degree of naturalness". Many authors have used this concept also in the field of LCA (see Brentrup *et al*. 2002 for a review) to classify different types of land, with the implicit assumption that the more natural a piece of land is, the better.

[8] This method updates and greatly improves a previous version of the same authors, the EcoIndicator 95. Therefore, no reference is done of the former method.

[9] Possibly, the world's most widely used LCA software.

Guinée *et al*. 2001 (CML 2001)

Representation	Only competition included. They refer to Lindeijer *et al*. (1998) and Köllner (2000) for further details on the impacts on the biodiversity and life support functions.
Mechanism & Model	Quantification as A * t, with no regionalisation, and no specification of land use types. Direct link between $m^2 \cdot y$ occupied and competition assumed.
Choices & Assumptions	All areas used cause the same competition
Applicability	Direct from the LCI results (as long as these express the land use for inventory items).
Conclusions	Does not directly consider occupation impacts and effects on life support functions. It may be enough as a first rough approximation, although it is not very sensitive for the comparison of similar systems, and does not allow for the assessment of maintaining types of land uses.

Weidema & Lindeijer 2001 (LCA GAPS)

Representation	Mainly deals with the occupation impacts related to the endpoints "productivity" and "biodiversity". Biodiversity includes scarcity on the ecosystem level.
Mechanism & Model	The impacts of occupation are calculated as the area affected times the duration of the occupation (including the relaxation period) times the difference between the current level of quality and the reference state: $I_{occ} = A \cdot t \cdot (Q_{pot} - Q_{act})/s_i$ (s_i is a slope factor for the relaxation time). Net Primary Productivity (NPP) is used as a quality indicator for productivity, whereas a biodiversity indicator is developed with regional data including species richness, inherent ecosystem scarcity, and ecosystem vulnerability.
Choices & Assumptions	Similar to Lindeijer *et al*. (1998).
Applicability	Potential vegetation type needs to be known, and only for a few land use types are % of original species loss known. The authors give general data for the main biomes.
Conclusions	This method represents actually a further refinement of that presented by Lindeijer *et al*. (1998). In this case, a more complex and comprehensive indicators is used for biodiversity, and NPP is chosen instead of fNPP for life support functions; the reason explained by Weidema & Lindeijer (2001) is that the latter is more adequate for the nature development, rather than the general life support (understanding life as nature and humans). Nevertheless, this renders the indicator less suitable for comparing continuing land uses (e.g.: different agricultural systems may have similar NPP, but one returns more vegetable residues to soil…). The same drawbacks apply on the usefulness of growth indicators for soil quality.

The main characteristics of the above-mentioned methods are highlighted in Table IV-4. Apart from the endpoints that are represented by each method, a consideration is done on whether the methods allow for comparisons between systems that are not transforming land, i.e.: maintaining types of land use. Again, this consideration is made based on the impacts on life support functions.

Table IV-4: Representation and allowance for comparison between maintaining types of land use in current LCIA methodology.

Reference	*Representation*			*Allows for comparisons between maintaining types of land use?*
	Biodiversity	*Life support functions*	*Occupation*	
Heijungs *et al*. 1992	✓ Qualitative: land use types	✗	$m^2 \cdot yr$	✗
Fava *et al*. 1993	✗	✗	m^2	✗
Cowell 1998	✓ Physical Habitat Degradation	✓ Organic Matter + Soil Compaction	no reference to the relaxation time	Assuming that any addition of Organic Matter is positive, and that machinery use and compaction are linear
Mattsson *et al*. 1998	✓ species richness and aesthetic value of landscape	✓ several indicators not aggregated	$m^2 \cdot yr$ no reference to the relaxation time	Uses qualitative and quantitative information, which does not allow for direct comparisons unless some valuation is done
Müller-Wenk 1998	✓ threatened vascular plant species	✗	$m^2 \cdot yr$ including a re-naturalisation time	✗
Lindeijer *et al*. 1998	✓ vascular plant species richness, α	✓ fNPP	$m^2 \cdot yr$ · [quality measure: α or fNPP]	Assuming that any addition of fNPP to the system is enough to keep soil quality and ecosystem's functions
Baitz *et al*. 1999	✗	✓ several indicators not aggregated	$m^2 \cdot yr$ · [quality measure for each indicator]	Yes, but a subjective assessment of the fulfilment of different functions is needed
Goedkoop & Spriensma *et al.* 1999	✓ Potentially Disappeared Fraction (PDF)	✗	$m^2 \cdot yr$ including a recovery time	✗
Guinée *et al*. 2001	✗	✗	$m^2 \cdot yr$	✗
Weidema & Lindeijer 2001	✓ indicator including α, inherent ecosystem scarcity, and ecosystem vulnerability	✓ NPP	$m^2 \cdot yr$ · [quality measure: biodiversity or NPP]	✗ (yield is not related to the maintenance of potential life support functions)

✓: included / addressed.
✗: not included.

When dealing with land use impacts in LCIA, most authors suggest indicators related to biodiversity and the "degree of naturalness" of the system. Only in the last years have indicators related to the life support functions of land been suggested (Cowell 1998; Mattsson *et al*. 1998; Lindeijer *et al*. 1998; Baitz *et al*. 1999; Weidema & Lindeijer 2001). Besides, land use impacts in LCA have been mainly considered from the point of view of changing land uses (e.g.: from forested area to arable land, from grassland to quarry, etc.), and no attention has been paid to the impacts of actual occupation. When occupation impacts are addressed, most of the methods suggested for land use impact assessment are not very precise in the sense of being able to detect small changes. Indeed, the quantification of organic matter added to soil suggested by Cowell (1998) is not enough as an indicator of the effects on soil quality, because it represents only one of the parameters affecting the organic matter balance in soil, which in turn affects soil quality.

Opposed to this reality, the relatively small occupation impacts arising from agriculture need to be assessed in order to determine the environmental preference of one system over another. The reason is that a big part of the impacts produced on land come from agricultural land (i.e.: land that is not being transformed), which continues under the same land use and yet important (occupation) impacts occur. Müller-Wenk (1998) determines that important effects on biodiversity are caused by land use even if no transformation of natural areas occurs. Mattsson *et al*. (1998) cite that 640,000 km^2 in Europe and the European part of the former USSR are degraded due to agricultural mismanagement, without actually changing the type of land use.

As LCA is increasingly being used as a tool for the assessment of agricultural practices, a proper assessment of the impacts derived from agricultural land use is needed. Indeed, the impacts on land are one of the main differences between different types of agriculture (see e.g.: Annex I of the European regulation on organic agriculture CEE 2092/91), and so they should be thoroughly assessed in order to be able to compare them. Moreover, the impacts on soil quality should be measured by soil indicators, as yield related indicators may not be good representatives of the long term effects on soil, and consequently on the ability of land to support life (see the discussion above and the references: Burger & Kelting 1999; Bouma 2002).

IV.2. Life support functions of land: soil quality degradation and assessment

This section explains the life support functions of land, which are mainly related to soil quality. Therefore, a convenient definition of soil quality based on life support functions is given. Then, the main processes of soil quality degradation are discussed with a specific emphasis on the main types of agricultural soil degradation, as well as their effects on agricultural systems environmental quality. Finally, a general overview of the approaches used in fields of research other than LCA for the assessment of land and soil quality is given.

IV.2.1. Life Support Functions and soil quality

Life support functions (LSF) concern the major regulating functions of the natural environment, which enable life on earth (both human and non-human). According to Udo de Haes & Lindeijer (2002) these can be summarised as:

1. **Regulation of earth climate**. Earth climate is mainly regulated by air emissions (which are not necessarily linked to land use), but the influence of land use on climate change is increasingly acknowledged. This relationship is ruled by the role of soil as a source or a sink of carbon (EEA & UNEP 2000). Also the effects of different land uses on N_2O emissions are of concern when talking about climate change. On a local scale, the vegetation cover of land also influences climate (by changing evaporation and transpiration, the solar heat absorption, wind speeds…), and vegetation cover is also influenced by soil properties (and vice versa). This local climate effect has not received much attention until now in LCA, and will not be a major matter of discussion in the present thesis (see section IV.3.7 for some brief considerations). It is important to note, though, that climate regulation by land is controlled through soil processes.
2. **Maintenance of substance cycles** (chiefly water, carbon and nutrients). In the case of substance cycles, their maintenance includes the position of land in the freshwater and the carbon (energy) circuits, the filter and buffer functions (sink of pollutants), and the microbiological transformations leading to the regeneration of nutrients. It must be noted that most of these cycles occur in soil, and not on land surface. Of course, in the case of water also surface processes (infiltration, runoff) are crucial, but these are also influenced by soil structure and porosity.
3. **Biotic production** (soil fertility). Finally, the ability of land to produce biomass does not only depend on the amount (area) of productive land, but also on its intrinsic capacity to do so, which depends on the quality of the upper layer (the soil). This ability has been usually called soil fertility. Soil fertility is obviously affected by soil properties, mainly nutrient

content, *structure*[10] (allowing for root penetration) and water holding capacity (EEA 1995). Also the microbial activity and the erosion resistance of soil structure are crucial for soil productivity. In summary, soil fertility is usually subdivided in three aspects: chemical, physical and biological fertility.

As a conclusion, it can be stated that in a certain piece of land, the ability to perform the life support functions is mainly determined by the characteristics of its soil. Indeed, Burger & Kelting (1999) identify most of these life support functions when discussing what soils must “do” in order to be productive in a forest management scheme: support plant productivity (soil fertility), regulate (forest) hydrologic cycle, regulate carbon balance, and bioremediate waste products. Daily *et al*. (1997) focus on the services that soil provides to plants (mainly the ecological functions mentioned in Teller *et al*. 1995, see Table IV-3 in page 182), and thus indirectly to humans, through:

- Buffering and moderation of the hydrological cycle
- Physical support of plants
- Retention and delivery of nutrients to plants
- Disposal of wastes and dead organic matter
- Renewal of soil fertility
- Regulation of major element cycles

Soil is thus the key component of life support functions and determines to a great extent the uses that can be performed with a certain piece of land. This is thoroughly discussed in a special issue of Agriculture, Ecosystems & Environment (issue 88, 2002), which is the result of an international workshop on “Soil Health as an Indicator of Sustainable Land Management” (24-25th June 1999, Greece).

Defining soil quality

Defining soil is not easy, due to the complexity of the processes and components that form it. Many authors (see e.g.: Domènech 1997; Kelting *et al*. 1999) define soil as a mix of interacting processes (biological, chemical, and physical), or as an interface between different Earth’s components (biosphere, lithosphere, atmosphere, hydrosphere…). Also, the dynamic and living nature of soils and the variability in the composition of its mineral and organic components are highlighted. In Figure IV-6, some of the main interacting processes that have a role in soil formation are shown.

[10] Soil structure is defined by the way individual particles are organised in bigger units (aggregates) and the void space associated to these units (Porta *et al*. 1994).

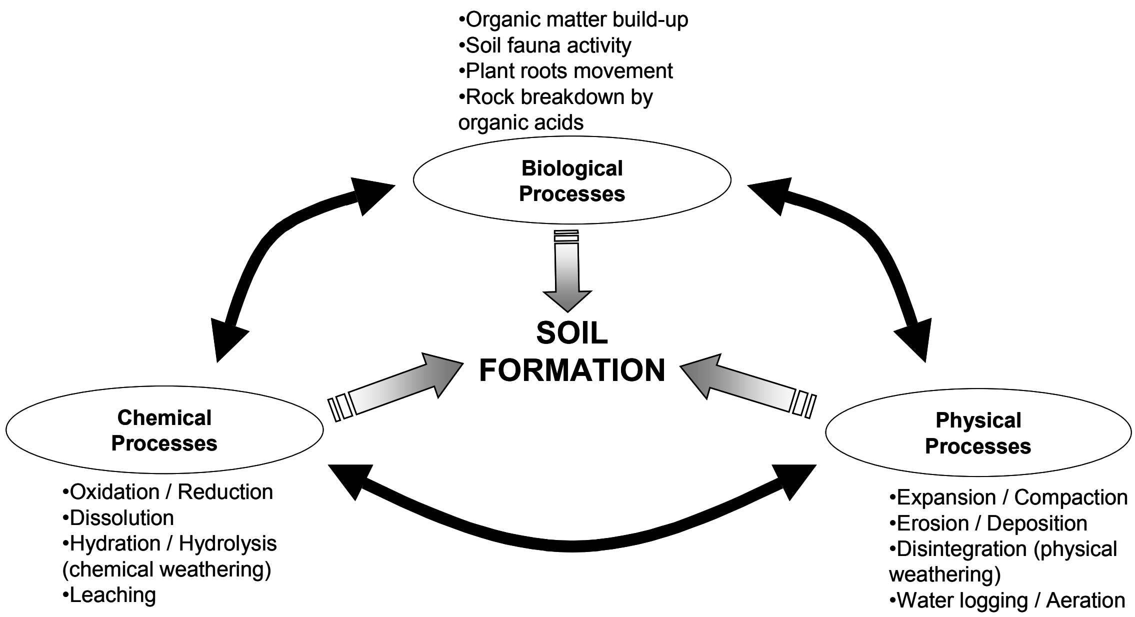

Figure IV-6: Some of the main soil formation processes.

Soil formation is the result of the interaction of all its components for decades to thousands of years. From this temporal point of view soil could be considered as a non-renewable resource[11]. Indeed, even though soil is not actually "consumed" (see Figure IV-4 in page 179), its loss through erosion or quality degradation can be considered as a depletion of the resource when no recovery of the quality occurs within the scope of the study (Lindeijer *et al*. 2002). Cowell (1998) used values of current loss and formation rates to calculate a static reserve life of ca. 104 years for soil[12]. It is calculated that in the United States soil has been lost at about 17 times the rate at which it has been formed (Troeh & Thompson, 1993), and this figure can be even worse in other parts of the world. Hence, soil use in modern world is unsustainable. The discussion on soil loss and formation is relevant for the soil depth, which is a possible indicator for soil fertility. However other aspects of soil quality are very important along with soil depth, and a broader perspective should be considered when assessing soil quality loos due to land use.

Soil quality may be defined as the capacity of soil to function (Karlen *et al*. 1997). This can be related to the ability of soil to perform the life support functions, which depends on physical, chemical and biological parameters. More precisely, Doran & Parkin (1994) have defined soil quality as "the capacity of a soil to sustain biological productivity, maintain environmental quality, and promote plant and animal health". Most authors dealing with soil quality relate it to the function of biotic production because they come mainly from the fields of agronomy and/or forestry, where production is the key issue (see e.g.: FAO 1979). In every situation, though, the land "user" may have different objectives and thus the different functions of land and soil will be attributed different importance (Karlen *et al*. 1997; Kelting *et al*. 1999). Also the concept of soil resilience is important in this context. Resilience has been defined in ecological science as the ability of a system to

[11] A non-renewable resource is that with a formation rate slower than the consumption rate. In the case of soil, there is no consumption as such, but it is recognised that degradation rates are relatively mush faster than formation and regeneration rates (COM 2002).

[12] Note that static reserve lives may vary greatly depending on local conditions.

return to a dynamic equilibrium after a disturbance. Therefore, the factors that make a soil more resilient also add to the quality properties, as they assure the performance of soil functions after degradation. Lal (1998a) defines soil quality as the net effect of the difference between resilience and degradation.

According to Brady & Weil (1999), soil quality reflects a combination of physical (*texture*[13], structure), chemical (composition) and biological (organic matter, biota…) properties, which are the result of the interacting processes defining soil (see Figure IV-6). While texture and mineral composition are relatively stable, structure and soil organic matter can significantly be changed by land management, and can thus indicate the status of a soil's quality relative to its potential (Brady & Weil 1999). Figure IV-7 has the same structure as Figure IV-6, because the soil formation processes define soil properties relevant for soil quality. Accordingly, a "good" soil (with high quality) may be defined as a "well-formed soil". Soil organic carbon (SOC) should be classified as a soil chemical property, but in Figure IV-7 it is considered a biological property because of its origin.

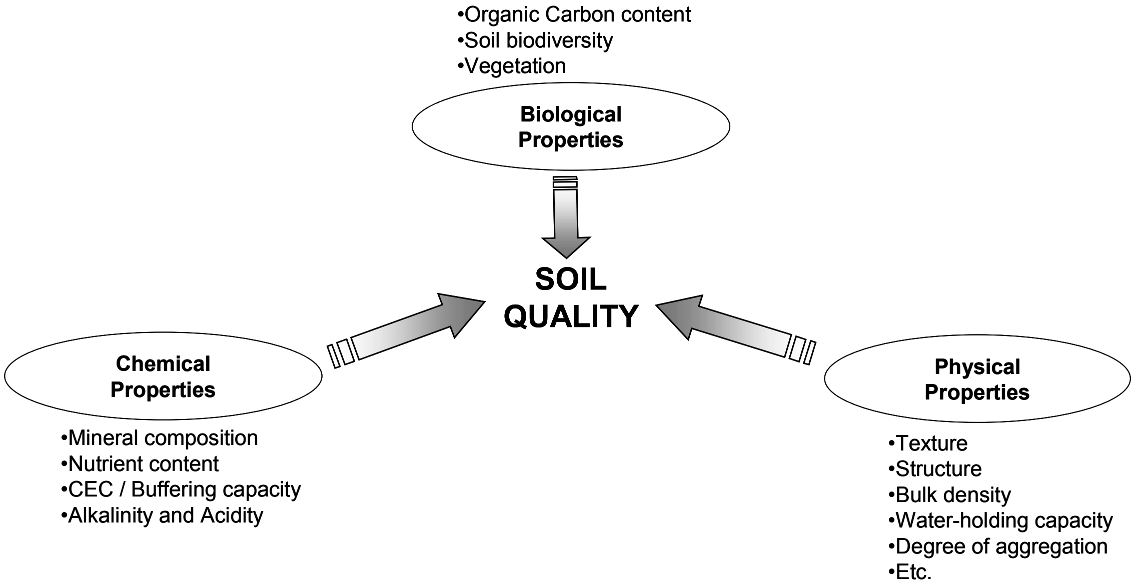

Figure IV-7: Properties affecting soil quality.

IV.2.2. Soil degradation

As it has been stated above, soil is multi-functional, which makes it important for many purposes, notably for food production. Also, this makes it vulnerable[14] from many sides (EEA & UNEP 2000), and damage to soil is not easily recoverable. On the other hand, damage to soil is not readily

[13] Soil particles are classified in several fractions according to their size: sand, silt and clay. The different proportions of these fractions define soil texture.

[14] Soil vulnerability to degradation is defined as the capacity of the soil system to be harmed in one or more of its ecological functions (Teller *et al*. 1995). It is noticeable that this definition considers only ecological functions of soil (in general, the life support functions) to be vulnerable; socio-economic functions such as foundation for buildings and infrastructure are less demanding in soil quality, and thus less vulnerable. Only

perceived due to its resilience and buffer capacity, which means that soil degradation is not often noticed until it is far advanced (EEA & UNEP 2000; Teller *et al.* 1995). The three main types of soil degradation according to their origin (Moss 1987) are:

- pollution via vegetation cover (atmospheric- and land-based)
- agricultural practices (physical and chemical effects[15])
- urbanisation and industrialisation (direct loss and sterilisation)

Only the two last types are actually linked to land use related impacts. Surface sealing by urbanisation, industrialisation, and infrastructure construction represents an irreversible loss of soil (depletion) through a permanent change in quality (transformation impacts). In the case of agriculture, the impacts may not be so evident, but the fact that they affect large areas of the most productive land makes them very relevant for sustainability.

Agricultural soil degradation: impacts on Life Support Functions

As mentioned above, the life support functions of land depend mainly on soil quality, and that is why in policy-making documents the stress is put in the need for protecting soil quality, rather than land (Moss 1987; COM 2002). Hence, the impacts affecting the performance of these life support functions are commonly known as soil degradation. When we talk about soil degradation we usually refer to the deterioration of its functioning. This degradation can be irreversible (e.g., when soil is lost through erosion and so its functions are no longer available) or partially reversible (e.g., soil is polluted, or nutrients are temporarily exhausted).

Agriculture is one of the main sources of impacts on land. The agricultural impacts on soil quality are generally reversible and thus can be recovered naturally or with the aid of some human action. The transformation process from natural land (forest, grassland) to agricultural land can thus be generally reversed with no transformation impacts. Beaufoy (2001, p. 33) comments on the case of olive farming in Spain, and concludes that “except in very adverse conditions, abandonment [of olive groves] without tree removal tends to result in scrub invasion and the gradual development of natural woodland, which provides a high level of soil protection…”. This is thus a typical case of re-naturalisation, although he also cites that some minor human intervention is advisable in order to reduce the risk of further degradation. Of course, examples of irreversible changes induced by agricultural practices also exist[16]; these generally represent the effects of keeping for a long time a slow soil degradation rate that was reversible until a critical point was reached.

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extreme conditions of location or severe land degradation (contaminated sites, serious erosion…) make it impossible for soil to develop socio-economic functions.

[15] Also biological effects are important, as will be noted below.

[16] Gardner (1998) mentions the case of the North of Africa, where two thirds of the cereals consumed by the Roman Empire in Southern Europe during the first century were produced. The organic matter and nutrients exported with this one-way flow did not return to Africa, and by the half of the third century the region started an ecological and economical decline. Also the case of The Dust Bowl in the USA Great Plains is a classical example of such irreversible changes. Jackson (2002) reviews many of the cases of unsustainable soil use

Most agricultural impacts on land are derived from the occupation (i.e.: occupation impacts). Also, soil quality may be declining during the occupation due to the human activities (as shown by the declining curve between t_1 and t_2 in Figure IV-4, page 179). This change is in general very slow (Sands & Podmore 2000) and difficult to detect in the short term, partly due to the above-mentioned soil's resilience. It is this slow change in soil quality that differentiates some major agricultural systems, and so it should be included in agricultural LCIA in order to be able to distinguish between these systems.

According to Lal (1998a), the impacts caused by agricultural activities on soil can follow physical, chemical, and biological mechanisms (see Figure IV-8). Again, Figure IV-8 shows a parallel structure to the previous figures (of soil formation processes, see Figure IV-6, and the properties defining soil quality, Figure IV-7). The obvious reason is that the same soil components defining its quality may be degraded by mechanisms that are similar to the soil formation processes. Indeed, some of the mechanisms mentioned in Figure IV-8 are actually the reverse of the soil formation processes shown in Figure IV-6.

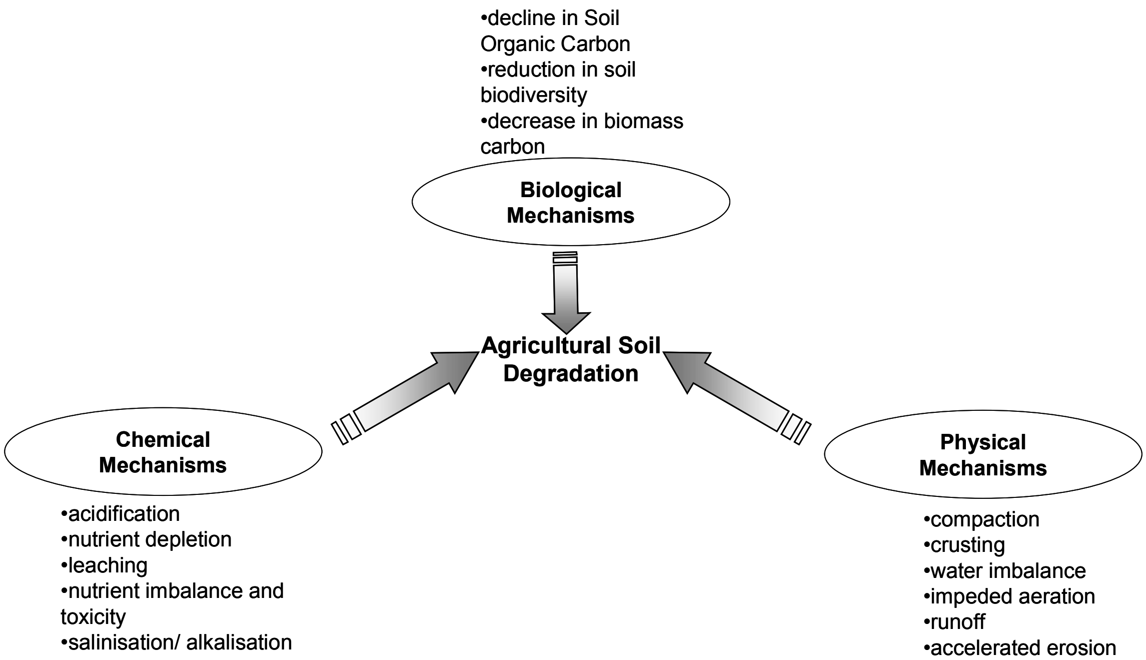

Figure IV-8: Agricultural soil degradation mechanisms (adapted from Lal 1998a).

An indicator that tells us about the degradation of soil's ability to perform the life support functions is thus needed. Some indicators have been suggested until now for this purpose, mainly focused to represent the biomass productivity of soil. Net Primary Productivity (NPP) or free NPP (fNPP) has been suggested as a single indicator (Lindeijer *et al*. 1998; Lindeijer 2000b; Weidema & Lindeijer 2001) due to its relationship with substance (mainly carbon) cycles and the biomass production (soil fertility) of an ecosystem. Cowell (1998), Baitz *et al*. (1999) and Mattsson *et al*. (1998; 2000) present combinations of indicators that are related to the different life support functions (including

by agriculture in the past 10,000 years, concluding that most civilisations failed to hold the topsoil, and that was the main cause of their decline.

and/or stressing biotic production), even though the practical implementation and weighting of such combinations is not clear.

IV.2.3. Existing types of soil quality assessment

Land and its soil are cornerstones of human development, and therefore we should exercise a use, and not an abuse, of them. For the reasons pointed out above, the effects of land use on soil quality should be thoroughly assessed in order to prevent further degradation and assure the environmental sustainability of human development. With this purpose, different methods for soil quality assessment have been developed in the last decades, which can be broadly classified in two main groups:

1. Large scale, low precision: global/regional maps by e.g.: FAO (Food and Agriculture Organisation), UNEP (United Nations Environment Programme), the EU (European Union) CORINE information system.
2. Small scale, high precision: experimental plots, soil fertility models, etc. for determining the influence of soil fertility factors on crop production.

Karlen *et al*. (1997) further define the scales of soil quality evaluation (see Table IV-5).

Table IV-5: Scales for soil quality evaluation (adapted from Karlen *et al*. 1997).

	Scale	Research level	Uses
MONITORING SOIL QUALITY	4. Regional, national, international	Policy development	Sustainable resource use decision-making
	3. Farm / Watershed	Inter-agency monitoring	Land use planning
	2. Field / Forest	Interdisciplinary; involving land managers	Select practices that enhance soil quality
UNDERSTANDING SOIL QUALITY	1. Plot	Disciplinary applied research	Soil quality changes with management practices
	0. Point scale	Sub-disciplinary basic research	Attributes and indicators of soil quality

Large scale assessment methods: land capabilities and vulnerability

Assessment at levels 3 and 4 is usually done with the aim of determining land capabilities and land vulnerability. The main idea behind these methods is the monitoring of vulnerable areas, and advising with general practices useful for national or regional agencies (level 2).

Some major works can be cited in this group: FAO (1979); the GLASOD project (Global Assessment of Soil Degradation) by ISRIC & UNEP (1991); the CORINE project (EEA 1995). They are focused at the assessment for decision-making at big scales (national or global levels), and changes due to specific management practices of specific systems are not of concern. The classification of land types is made according to general land uses, which broadly define the capabilities of the land (forest, arable, desert, etc.). Also the state and/or risk of degradation are

usually included in these studies leading to improved land use planning and management (ISRIC & UNEP 1991).
Generally, these methods aim at obtaining maps of degradation potentials and/or agronomy capabilities. The USDA[17] produced many works on these issues (see Porta *et al*. 1994, p.560-571), including the land classes classification, and the “prime farmlands” and “unique farmlands”. The parameters assessed in these studies include climate, topography, and soil characteristics.

Small scale assessment methods: soil quality

The second group includes studies focused on few soil parameters and their effect on crop productivity and the life support functions of soil. They are usually developed for the assessment of agricultural soil and practices and the understanding of soil’s complex processes. It is often difficult to draw conclusions from them, as it is nearly impossible to control all the factors that determine crop productivity.
As soil quality is defined by the performance of soil’s functions, soil quality evaluation needs to take this performance into account. As stated above, soil is complex and its quality is multi-functional. In order to address the difficult task of assessing this complexity and multi-functionality, the functions that need to be assessed and/or protected must be carefully defined. For instance, soil productivity indicators are usually defined, as productivity is generally the goal of these assessments. Then, soil attributes affecting these functions must be identified, and finally a minimum set of indicators should be selected in order to measure those attributes (Kelting *et al*. 1999).

Soil quality assessment for agricultural LCA

This second group of small scale, more detailed, soil quality assessment methods is the relevant one to include soil quality assessment (i.e.: impacts on life support functions) in LCA that involve agricultural production. The reason is that agricultural practices will determine to a great extent the diminution/change in soil quality after its use by the system. Soil quality indicators will generally be soil- and site-specific (Burger & Kelting 1999), and this group of soil quality assessment methods allows for the site-dependency that is needed in agricultural LCIA. Also, soil-related problems are site-specific, which makes any attempt of generalisation very difficult (Teller *et al*. 1995) and requires a case-by-case application.
Indeed, it was concluded in section III.5.1 that soil type may be one of the main factors determining the environmental interventions of an agricultural system. Besides, in section III.5.2 it is suggested that land use related impacts should be considered from a site-dependent perspective. This is reinforced by the LCIA framework for land use impacts, explained in section IV.1.2, where it is pointed out that land use impacts are always referred to a reference state, which obviously depends on the local conditions. Furthermore, as discussed in the conclusions of section IV.1.4, most currently available methods for the assessment of land use impacts in LCIA are not suited to assess occupation impacts, and are therefore quite useless when comparing agricultural systems. Agriculture-induced land degradation occurs at the soil point scale, and therefore an indicator developed at this scale is needed (see Figure IV-3 in page 177 and Table IV-5).

[17] United States Department of Agriculture.

IV.3. Soil Organic Matter as an indicator for impacts on life support functions due to land use in agricultural LCA

This section suggests a new indicator for the assessment of impacts on the life support functions of land in agricultural LCA based on the Soil Organic Matter content. The scope of this indicator is the evaluation of long-term impacts caused by the continuous occupation of land by agriculture. Apart from the impacts on biodiversity, the main effects of agriculture are on the occupied soil, and so, as it has been explained in section IV.2, on life-support functions. Other impacts, caused off-site (e.g.: eutrophication by leaching nitrates, toxicity from pesticides and heavy metals, etc.) are considered under other impact categories.

Soil Organic Matter (SOM, from now on) has already been suggested as an indicator for soil quality in previous LCIA methodologies, generally as a measure of soil attributes to be combined with other parameters (such as structure, pH, rainwater infiltration, etc.; see Cowell 1998, Mattsson *et al*. 1998 and Baitz *et al*. 1999). Nevertheless, methods that rely on different indicators need frameworks to aggregate them, which are not generally addressed. Here, it is argued that SOM can work properly as a single indicator for life support functions in agricultural soil, and a way to implement it is discussed. SOM is probably the most cited indicator of soil quality (Allison 1973; FAO 1979; SSSA 1987; Barrow 1991; ISRIC & UNEP 1991; Karlen *et al*. 1997; Lal 1998a; Stenberg 1998; Sands & Podmore 2000; Arshad & Martin 2002; Nortcliff 2002). Brady & Weil (1999) relate it to the dynamic nature of soil, which differentiates it from abiotic resources and gives it the added value to support life. This is because SOM is the result of biological activity (soil organisms and vegetation) and so it differentiates weathered rock (affected by physical and chemical processes) from soil. Furthermore, Brady & Weil (1999) state that enhancing the quantity and quality of SOM is a central factor in improving soil quality. Additionally, Reeves (1997) gives possibly the best review of SOM as a soil quality indicator, and concludes “soil organic carbon (SOC) is the most consistently reported soil attribute from log-term studies and is a keystone soil quality indicator”. Actually, it is not only linked to soil quality, but also to soil resilience[18] (Lal 1998a): a soil with high SOM is less vulnerable. Accordingly, the European Commission exhorts that any future policy on soil protection shall warrant the protection of soil biodiversity and SOM, as “these are the keystones for soil functions” (COM 2002).

As it has been explained in section IV.1.1, an indicator should comply with several characteristics in order to properly assess an impact (see Table IV-1 in page 177): representation, measurability, consistency, applicability, site-dependency, scale, and transferability. The following sections describe the appropriateness of SOM as an indicator for the life support functions of soil in relation to these characteristics. Accordingly, section IV.3.1 reviews the references discussing the relation between SOM and life support functions, with the aim of determining the degree of representation of life support functions by SOM. Section IV.3.2 presents the ways of obtaining data on SOM levels and gives reference values for different soils and world regions. Then, section IV.3.3 deals with the

[18] Soil resilience is the inverse of soil vulnerability.

methodological implications of applying the indicator, from the point of view of the LCIA framework. In addition, the consistency of the indicator with the LCIA framework is discussed in section IV.3.4, with a special focus on the issues of site-dependency and the appropriateness of the indicator's scale in relation to the problems that need to be assessed. Section IV.3.5 evaluates the indicator according to international standards, and section IV.3.6 deals with the allocation procedure for land use impacts among successive land uses. Finally, the relation of the SOM indicator with global warming is discussed in section IV.3.7.

IV.3.1. The role of Soil Organic Matter in Life Support Functions

Soil Organic Matter (SOM) can be defined as a complex heterogeneous mix of organic materials naturally present in soil (Porta *et al*. 1994). Generally, fresh organic matter (vegetable biomass, dead or alive) and microbial biomass are not included in this definition. The origin of SOM is diverse, most of it coming from dead biomass either naturally entering the soil (litter) or provided by humans in croplands (manure), the physical (mechanical) and chemical degradation of these organic tissues, and the synthesis of new complex compounds by soil micro-organisms (Porta *et al*. 1994). Even though SOM represents only 1-6% of productive arable soils' mass, its importance for soil fertility and other life support functions of land is crucial. Evidence of the high relevancy of SOM for the life support functions of land is given in the following paragraphs, and Figure IV-9 gives a graphical representation of the influence of SOM on life support functions.

Figure IV-9 has been adapted from Lindeijer *et al*. (2002) by keeping the structure for the representation of the cause-impact network of resources and land use, while giving special attention to the adverse effects on life support functions. Also, the relationships between soil quality and life support functions are highlighted, and the role of SOM in soil quality is specially addressed. It must be noted that Lindeijer *et al*. (2002) give a special relevance to biodiversity as "impact midpoints" affecting the endpoint life support functions, with a general reference to "soil degradation" as another impact midpoint affecting life support functions. No direct mention to SOM appears in their cause-impact network, though the free Net Primary Production is included as "(free) Biomass Production". The free Net Primary Production (fNPP) is closely related to SOM, as will be explained below (see Figure IV-12, in page 213).

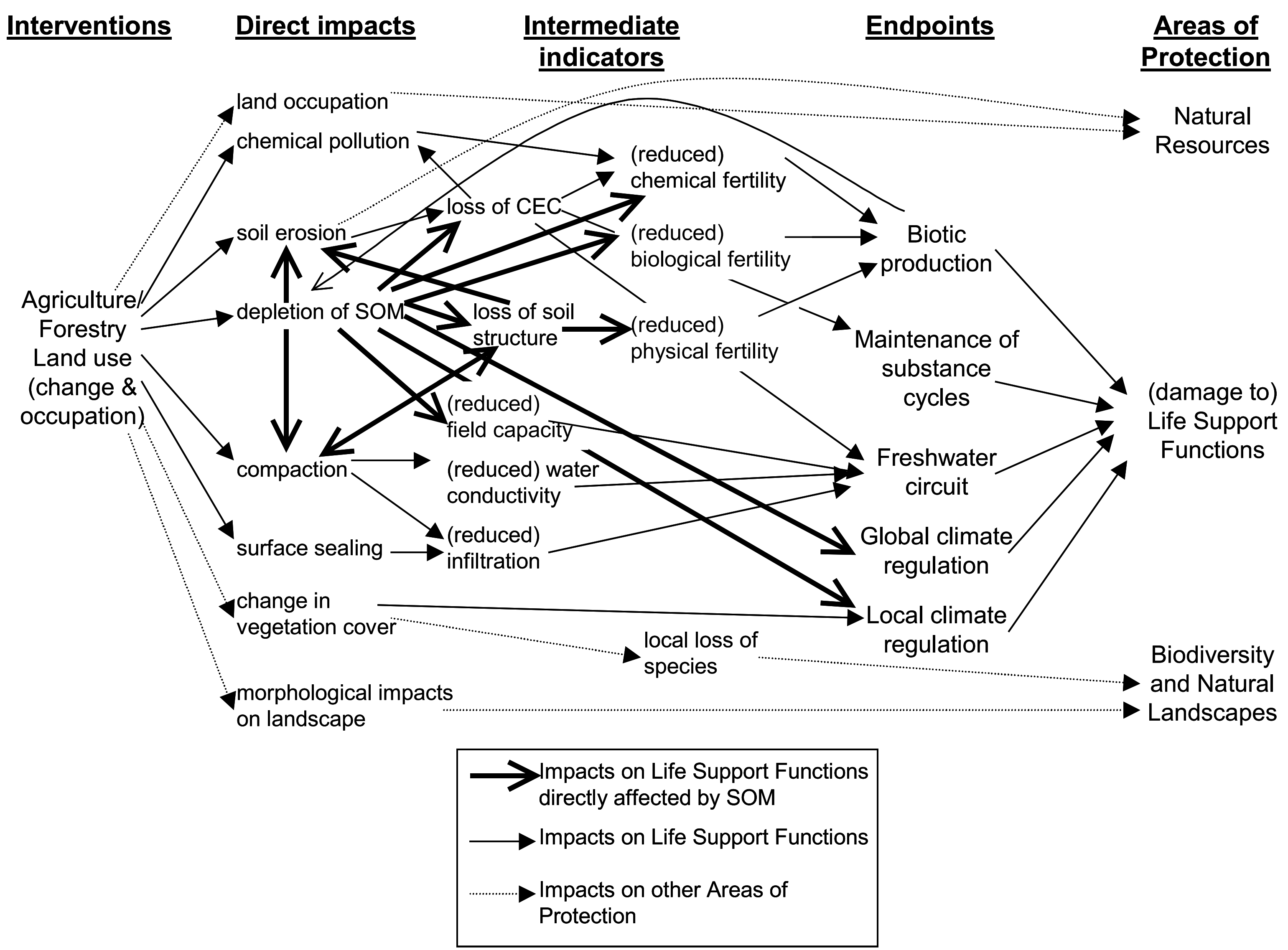

Figure IV-9: Cause-impact network of the impacts on life support functions caused by agricultural land use.

It is noticeable from Figure IV-9 that SOM is indeed related to most life support functions, either directly or indirectly. Note also the existing feedback between biotic production and SOM: the higher the biotic production, the higher the biomass return to soil and so the higher the SOM level (unless all biotic production is extracted from the system). This is why it is suggested that fNPP (the biomass that may be returned to soil) might be a proxy to SOM. In the same way, if SOM is depleted and soil fertility decreases, biotic production also decreases and so the biomass return to soil is reduced (while biodegradation goes on), with the additional decrease in SOM level.

Biomass production (chemical, physical, biological soil fertility)

Soil provides nutrients, air, water and a medium in which roots can penetrate (physical and chemical fertility), and SOM is responsible for most of this. SOM also enhances biological activity in soils (biological fertility).

There are many studies assessing the central role of soil organic matter in soil productivity and the evolution of soil organic matter under different cropping systems. Allison (1973) discusses thoroughly on the roles of soil organic matter in relation to crop productivity. In SSSA (1987), an overview is given on the relationship between soil organic matter and soil fertility. For instance, Cole *et al*. (1987) describe an experiment in Michigan that shows an increase of 20% in maize production for every 1% of increase in soil organic matter. Some long-term experiments describe similar results. Stenberg (1998) finds that other single parameters of soil, such as total nitrogen, are better related to soil fertility than organic carbon (or SOM); nevertheless, SOM is also an important source of N (around 90%). Actually, Stenberg finds in the same study that not single parameters, but multivariate models can better predict soil fertility; the best multivariate models are dominated by variables associated with the organic matter. Aune & Lal (1995) find that the relationship for the productivity response to changes in soil organic carbon is a positive increasing curve.

In relation to the chemical fertility, it must be noted that SOM is the primary sink and source of plant nutrients in terrestrial ecosystems (Paul & Collins 1997). The slow degradation of SOM through microbiological activity releases nutrients that can be absorbed by plants. Indeed, it provides most nitrogen (around 95% according to Paul & Collins 1997), 50-60% of phosphorus, and 80% of the sulphur needed by plants in an unfertilised soil (Domènech 1997). Besides, SOM acts as a buffer thanks to its Cation Exchange Capacity, slowly releasing nutrients and thus avoiding nutrient loss through leaching in phases when plants do not use them (see below). By attacking soil minerals with its acid compounds (humic and fulvic acids), SOM releases such cations as Fe^{3+}, Cu^{2+}, Zn^{2+} and Mn^{2+} and makes them available for plants.

SOM is also important for the physical fertility. The contribution of SOM to the soil structure helps in the formation of pores, which are used by plant roots to explore the soil depth; this penetration is also helped by the reduction in bulk density attributed to SOM (Díaz-Zorita & Grosso 2000; Mosaddeghi *et al*. 2000; Sharma & Bhushan 2001). In addition, these pores allow for air and water to circulate the soil space, and SOM increases the water holding capacity of soil and moisture retention (Sharma & Bhushan 2001), thus increasing the amount of water available for plants. Many authors have studied the positive effects of SOM in avoiding soil compaction (Soane 1990), which has long been recognised as a major factor affecting crop production. The effect is different under different soil conditions of texture, structure and moisture content (Zhang *et al*. 1997; Mosaddeghi *et al*. 2000), but the effects of adding OM are always positive in reducing soil's bulk density (an accepted indicator for soil compactibility). For instance, in a study on arable soil in Argentina, Díaz-Zorita & Grosso (2000) found that Total Organic Carbon (TOC) has a dominant effect on the susceptibility of soils to compaction. Smith *et al*. (1997) found that the effect of soil organic carbon content on compactibility increased in importance at low clay contents (<25 g 100 g^{-1}) in a study covering a broad range of forest soils. Quiroga *et al*. (1999) conclude that "decreases in OM content as a consequence of more intensive soil use (…) render soils more susceptible to compaction".

Organic matter supports soil organisms, which are crucial for soil biological fertility. As is further explained below, microorganisms degrade and recycle nutrients, making them available for plants. On a physical level, soil mesofauna (such as earthworms) maintains soil structure for crop growth (Benckiser 1997; Cerdà 2000).

Freshwater circuit

As for the relation of SOM with the hydrologic cycle, soil structure and porosity is responsible for rainwater infiltration, and, as noted above, the organic matter helps in providing such structure. Bruce *et al*. (1992) reported an increase in rainwater infiltration rate into soil related to higher content of SOM from crop residues. Water flow in soils is also related to soil compaction and thus the effect of SOM on compaction (see above) is also relevant here. Higher compaction and lower hydraulic conductivity are found to be related to lower SOM content by Quiroga *et al*. (1999) in Argentinean arable and virgin soils. Although it is not clear whether total organic matter or only fractions of it are responsible for aggregate stability (Dutartre *et al*. 1993; Adesodun *et al*. 2001), SOM is also one of the parameters that determine soil vulnerability to erosion through its role in soil stability (Falloon *et al*. 1998). Cerdà (2000) also finds a clear correlation between the amount of SOM and the soil aggregate stability, and recognises low SOM contents as a potential weakness against disrupting effects for aggregate stability such as tillage. Similar results are found by Saggar *et al*. (2001).

Filter and buffer capacity

The Cation Exchange Capacity (CEC) of a soil is the main parameter determining its buffer and filter capacity, and SOM is a major component (along with clays) of the CEC (generally accounting for 50-90% of the CEC in mineral soils, according to Brady & Weil 1999). Consequently, soils with higher SOM content are less vulnerable to soil pollution (Teller *et al*. 1995) because they can hold a higher amount of contaminants before they are saturated and start acting as a contaminant source. Also the importance of the CEC for the role of soil as a nutrient balance and consequently for chemical fertility has been mentioned (see above).

In addition, the fact that SOM is the major nutrient source for soil organisms makes it very important as well for the maintenance of soil biota, and thus the **gene reservoir** (genetic biodiversity) in soil. In this sense, SOM promotes life in the soil, and thus allows for the **degradation capacity** that is needed to deal with the harmful substances reaching the soil. The functional (economic) importance[19] of the gene reservoir function is highlighted by the fact that soil organisms are being studied that can degrade hazardous substances, in the field of bio-remediation. The degradation capacity is also crucial for recycling nutrients, as has been suggested earlier in this section. Soil micro organisms are even being studied for their potential application as biological control for weeds, thus reducing the need for other means of weed prevention such as synthetic herbicides (Quimby *et al*. 2002).

Climate regulation

In relation to the **climate regulation**, soil is a key component in the carbon cycle, which has been thoroughly studied in the last years as a result of the major concern about climate change (Falloon *et al*. 1998). The soils of the world contain three times as much carbon as in the above-ground living organisms, and twice as much as in the atmosphere (Teller *et al*. 1995; Falloon *et al*. 1998) (see Figure IV-10). The carbon pool in SOM is estimated at about 1400 Pg (Falloon *et al*. 1998; other references give values around this figure: Post *et al*. 1982 see Table IV-8; Eswaran *et al*. 1993 see Table IV-7). Even though climate is currently mainly affected by the imbalance from fossil fuels' carbon emissions, SOM can act both as a source or a sink for these emissions (CO_2 and CH_4). Hence, whether the soil acts as a carbon sink or a source will determine to an important extent the evolution of earth climate. According to Figure IV-10, soil carbon is being depleted (see that inputs from vegetation, 60 Pg/year, do not counterbalance the loss through oxidation, 62 Pg/year). Harrison *et al*. (1993) also suggested that soil carbon loss associated with agriculture is a significant source of atmospheric CO_2, estimating an annual contribution of 0.5 gigatons (Pg) of carbon during the 1980s. Nevertheless, Buyanovsky & Wagner (1998) find that this trend is changing due to the higher productivity in modern agriculture, and carbon is being sequestered by soil (around 32 Tg C annually in the USA during the last 40-50 years). According to these suggestions, then, most carbon lost from soil would be coming from land transformations, such as deforestation, mining or quarrying, etc. On the other hand, Larionova *et al*. (1998) suggest that the balance is neutral for cropland, while forest and meadow ecosystems' soils act as carbon sinks. Lal (2000) suggests that soil may sequester 0.9-1.9 Pg C yearly through desertification control and 3.0 Pg C year^{-1} through restoration of degraded soils in the world, while emissions from fossil fuels are 8 Pg C year^{-1}. In summary, it is not clear whether soil acts as a source or a sink for carbon, however it is obvious that whatever the trend will have a crucial influence on climate. Figure IV-12, in page 213, gives an overview of how agricultural management may affect the carbon balance in a crop field.

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[19] Obviously, the intrinsic value of biodiversity is also very important, but the area of protection "Biodiversity" covers this kind of values.

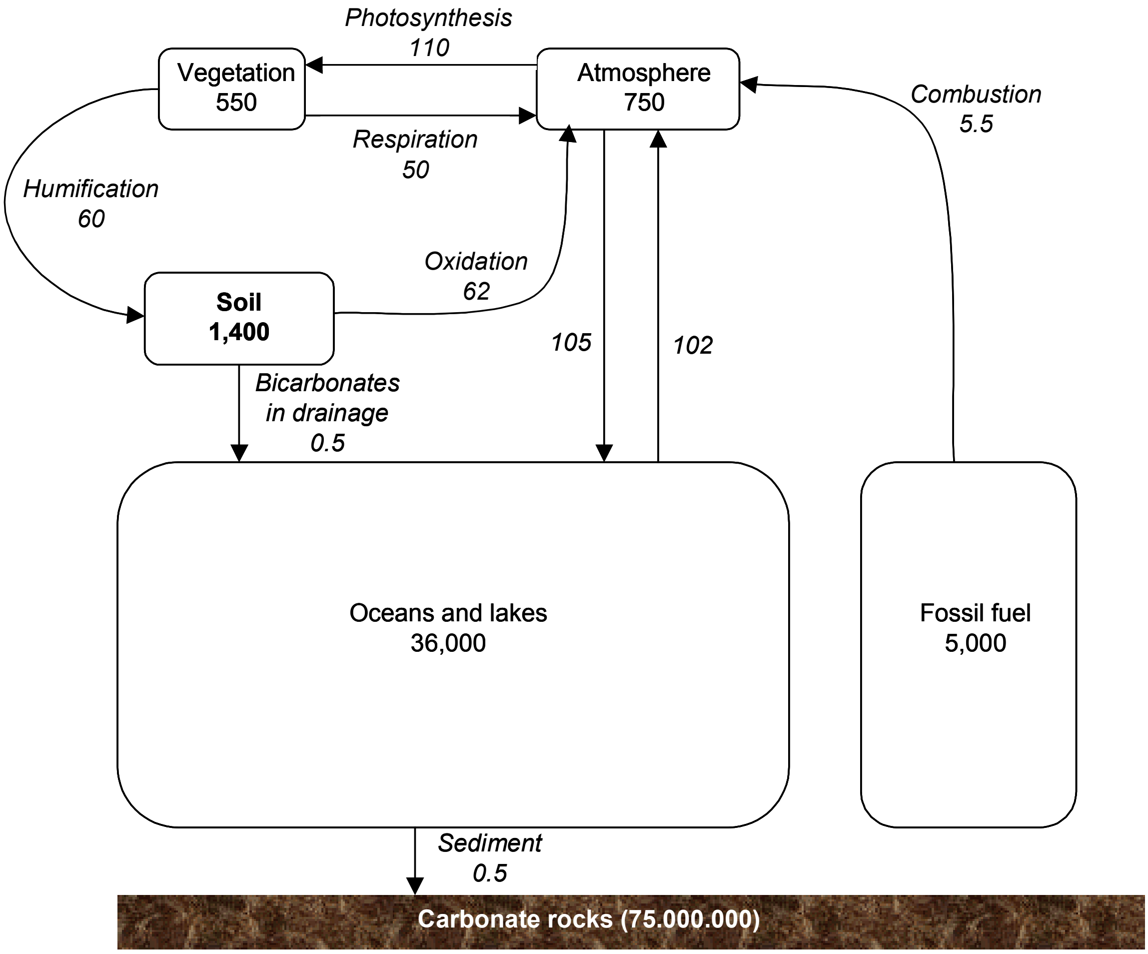

Figure IV-10: Global carbon pools (in Pg C) and *flows (in Pg C/year)* (Brady & Weil 1999, p. 447). The value for soil carbon pool has been corrected (in the original it is 2,400 Pg, but this contradicts all other references). 1 Pg = 10^{15} g.

Vegetation cover affects the local climate (precipitation, temperature…) through the albedo and surface roughness; as SOM is mainly a result of past vegetation, it can be linked to local climate as well. Also the darker colour of exposed soils with high SOM content reduces the albedo effect, thus affecting the local climate.

Table IV-6 summarises the role of SOM in the Life Support Functions:

Table IV-6: Role of Soil Organic Matter in the Life Support Functions.

Land function	*Role of SOM*
Maintenance of substance cycles	- **Freshwater circuit**: water-holding capacity; Cation Exchange Capacity (CEC) for its filter function; water conductivity; water infiltration - **Carbon cycle**: carbon pool - **Nutrient cycling**: microbiological activity - **Immission protection**: CEC (filter and buffer capacity of soil); degradation capacity (gene reservoir)
Soil fertility	- **Physical fertility**: soil structure (formation of aggregates) allowing for root penetration; contribution to erosion resistance and land stability; reduction of susceptibility to compaction; soil aeration - **Chemical fertility**: nutrient pool; nutrient protection (CEC holds nutrients avoiding their loss through leaching); pH control (buffer capacity); plant growth regulation - **Biological fertility**: enhancing soil biota (food source); nutrient cycling (degradation capacity and nutrient availability); microbial activity maintains soil's temperature
Regulation of climate	- **Global climate**: carbon cycle - **Local climate**: link to vegetation cover; reduction of the albedo of exposed soil

As a conclusion, it can be stated that soil organic matter is a representative indicator for the life support functions of land. It works following a positive curve with soil quality, i.e.: higher levels of SOM mean higher quality, while decreasing SOM is associated with soil degradation processes.

IV.3.2. Data sources for SOM

In general, the quantity of SOM in a system can be:

1. Measured directly from soil samples,
2. Calculated using models,
3. Estimated from literature values for different areas and crops.

Cowell (1998) suggests that these three options are actually hierarchical, and thus "real life" measures should be used whenever possible. Indeed, the uncertainty related to these sources of data will be much lower if real measures are available. Reducing the uncertainty in LCA results should be a goal in any agricultural LCA, as it was suggested in Chapter III, in order to increase the credibility of this tool for decision-making.

Soil organic matter can be expressed in many different ways (Lal 1998b; Nortcliff 2002). In practice, SOM content is usually estimated from the analysis of soil organic carbon (SOC[20]) content, because the latter can be determined more precisely and so it is generally used in scientific quantitative discussions (Brady & Weil 1999). Many studies exist that measure SOM as part of an experimental work on soil properties, or as a relation to productivity and/or other soil

[20] Brady & Weil (1999) suggest that SOM can be roughly estimated as 1.72 times SOC (i.e.: SOC represents ca. 58% of SOM).

functions. As a consequence, extensive datasets may be found in the literature on SOM levels in different soils of the world (see Table IV-7 for SOC in different soil orders, and Table IV-8 for SOC in different life zones). These datasets might be used as default figures for SOM levels, although SOM values will always be site-dependent, and consequently default values should only be used as a last option.

SOC or SOM levels are generally expressed in one of two ways: Mg ha^{-1} or %. In both cases, the depth of the measure needs to be known. When expressing SOC in % it refers to g C / 100 g soil.

Table IV-7: Mass of Organic Carbon in the World's Soils (Brady & Weil 1999; original data from Eswaran *et al*. 1993).

Soil order	*Organic carbon in upper 15 cm Range, % (g/100g)*	*Organic carbon in upper 100 cm Mg ha^{-1}*	*Global, Pg*
Entisols	0.06-6.0	99	148
Inceptisols	0.06-6.0	163	352
Histosols	12-57	2,045	357
Andisols	1.2-10	306	78
Vertisols	0.5-1.8	58	19
Aridisols	0.1-1.0	35	110
Mollisols	0.9-4.0	131	73
Spodosols	1.5-5.0	146	71
Alfisols	0.5-3.8	69	127
Ultisols	0.9-3.3	93	105
Oxisols	0.9-3.0	101	119
Misc. land	-	24	18
TOTAL			**1,576**

Table IV-8: Mass of soil carbon in world's life zones (Post *et al*. 1982).

Life zone groups	*Organic carbon in upper 100 cm Mg ha^{-1}*	*Global, Pg*
Tropical forest – wet	191	78.3
Tropical forest – moist	114	60.4
Tropical forest – dry	99	23.8
Tropical forest – very dry	61	22.0
Temperate forest – warm	71	61.1
Temperate forest – cool	127	43.2
Boreal forest – wet	193	133.2
Boreal forest – moist	116	48.7
Tropical woodland and savanna	54	129.6
Temperate thorn steppe	76	29.6
Cool temperate steppe	133	119.7
Tropical desert bush	20	2.4
Warm desert	14	19.6
Cool desert	99	41.6
Boreal desert	102	20.4
Tundra	218	191.8
Cultivated land	79	167.5
Wetlands	723	202.4
Global soil carbon pool		**1,395.3**

SOM or SOC values at more detailed scales are not generally available. France is possibly an exception to this, as this country is about to complete a National map of SOM covering the whole territory with a grid of 16x16 km (COM 2002), which will be updated every five years. In the near future, and at the European level, this availability of reference SOM levels is likely to increase (COM 2002).

Nevertheless, as explained in the following sections, not only the actual value of SOM is needed for the LCIA framework for land use impacts, but also the evolution with time of this parameter. In this sense, real data for time evolution of SOM will seldom be available, but models exist that can be used in order to calculate SOM evolution with time, and the effects of the system under study on this evolution. Powlson *et al*. (1996) give an extensive review of some of these methods, based on long-term datasets of SOM. Indeed, according to Bouma (2002) one major advantage of simulation models is that calculations can be made for many years, and thus the effects of weather and soil management practices can be provided. Accordingly, models can be used to predict the rates and direction of soil quality change (Arshad & Martin 2002). Obviously, the models are site-dependent, and therefore models that have been developed under similar conditions as those of the study are needed. The use of models is further explained in section IV.4.

IV.3.3. Applying the indicator

The feasibility of applying an indicator in a time- and data-consuming method such as LCA is generally related to data availability. As it has been pointed out above, even in the cases where real measures of SOM are not available, SOM models may be used in order to calculate SOM current state, the relaxation potential and the relaxation time, for a set of agricultural practices and site conditions. Alternatively, a set of average relaxation times for different agricultural activities could be generated in order to be used with the average relaxation potentials (SOM contents of different biomes, soils or land use types) given in Table IV-7 and Table IV-8; in this way, only initial and final SOM levels need to be measured / estimated / calculated. Nevertheless, this option seems unrealistic due to the many site-dependent aspects affecting the relaxation times and potentials.

SOM evolution under an agricultural land use

Generally, SOM losses caused by an agricultural land use will be fully reversible. The reasoning behind this assumption is that agricultural land use is actually exploiting the ability of land to produce biomass, and so this ability is not usually fully impaired. Therefore, even though SOM can be depleted to an important extent due to the imbalance between inputs and outputs during agricultural activities, natural vegetation can be re-established if land is abandoned with the consequent build-up in SOM. With this assumption, no transformation impacts are considered, and the relaxation time can be added to the total occupation time in order to account for the full occupation impacts (see Figure IV-11).

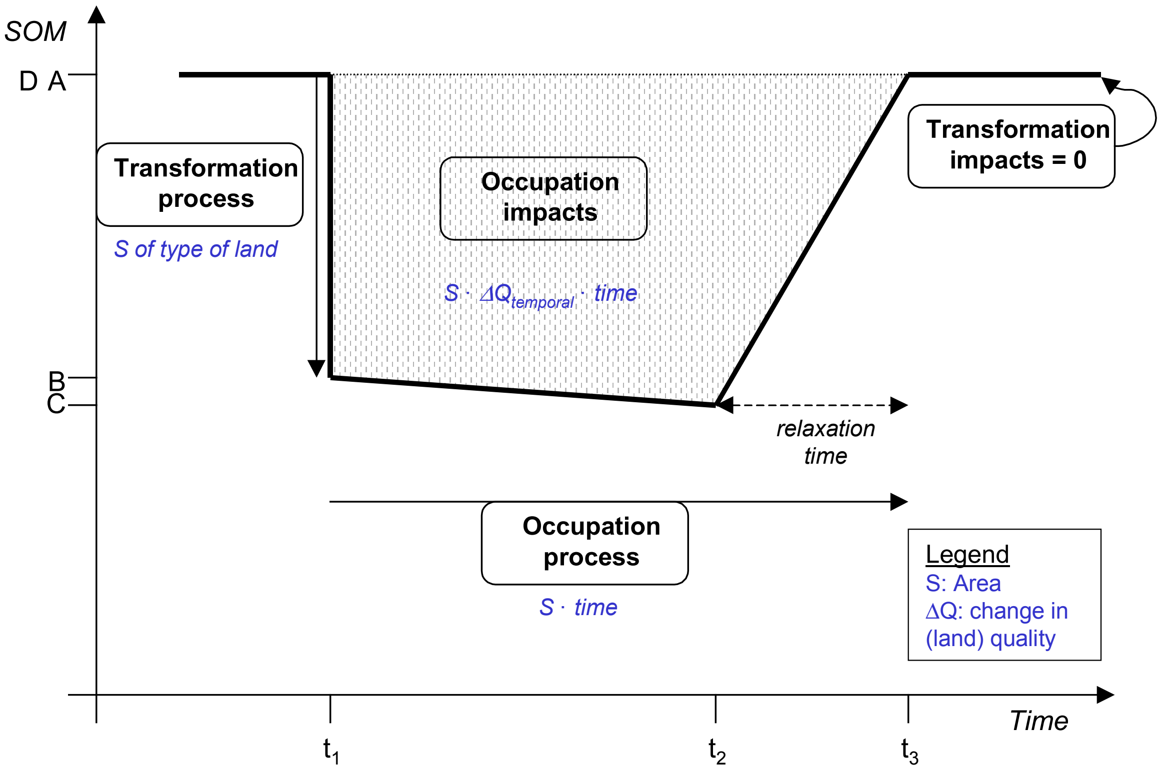

Figure IV-11: Evolution of SOM in an agricultural land use.

From a life support function point of view, it can be actually discussed whether transformations to agriculture do ever represent transformation impacts (net changes in the relaxation potential, as shown in Figure IV-4) at all. Even though the visual transformation is evident (there is a crop field where there was a forest, for example), the supportive capacity of soil is seldom handicapped, and so the potential for productivity and the other ecological functions is approximately the same. It is this potential for life support that should be protected, rather than the actual performance of life support functions (e.g.: actual productivity). It must be clear that this discussion only holds true for life support functions: impacts on biodiversity due to a change from forest or grassland to agriculture can be huge. Besides, it is only valid in some ecosystems (e.g.: in tropical rainforests, where most carbon is in the biomass and not in soil, when the trees are chopped down so is the life supportive capacity).

Calculation procedure

As shown in Figure IV-11 and discussed in Blonk *et al*. (1997), mainly three figures are needed for the assessment of the land use occupation impacts: the land occupation due to an activity (L_a) per functional unit (e.g.: ha·year/f.u.); the SOM value at the reference state (SOM_{ref}; A or D in Figure IV-11) and at each moment of the occupation process (SOM_a; from B to C in Figure IV-11). Then, the occupation impacts on life support functions due to an activity may be expressed as:

$$LSF_a = L_a \cdot \left(SOM_{ref} - SOM_a\right)$$

Once the value of SOM level during the occupation (SOM_a) is known, the SOM model may be used to predict the time that will be needed before the reference state (relaxation potential) is reached. Some expert judgement by ecologists on the most likely situation that may evolve after land abandonment will generally be needed to assess this evolution. Usually, the steady state that is reached after the land use has finished will be the relaxation potential for that site, because no transformation impacts are assumed in agricultural land use (see above). Otherwise, an average reference value of SOM_{ref} may be taken for the suitable biome or vegetation type from Table IV-7 or Table IV-8. Then, the values of SOM_a at each moment of the occupation process (including the relaxation time) can be also obtained from the model and subtracted from the reference value (SOM_{ref}). Multiplying each value (SOM_{ref} - SOM_a) by the calculation step of the model (e.g.: one year or one month) and the area of study, the occupation impacts are obtained.

Equation 1:

$$LSF_a = A_a \sum_{i=t_1}^{t_3} \left(SOM_{ref} - SOM_{a,i} \right) t_i$$

Where A_a is the area occupied by the activity and *i* represents the calculation step for which the model gives the results; the temporal limits of the calculation, t_1 and t_3, are taken from Figure IV-11.

Soil depth for the calculation and meaning of the indicator

The model results will generally refer to the density of SOC per unit area in a certain depth; the model user usually sets the depth. While Table IV-7 and Table IV-8 give the figures for the upper 1 m of soil, which generally represent most of the carbon in the soil profile, Brady & Weil (1999) suggest that the upper 15 cm represent the surface soil, which is most readily influenced by land use and soil management. Cowell (1998) suggests that the “furrow slice” (depth of soil affected by ploughing), which generally consists of the upper 30-50 cm of soil, should be considered when analysing soil degradation. Whatever depth is chosen for the analysis, it should be kept consistently throughout the calculations.

The results for SOM are thus obtained in kg C/m^2 in a certain depth, and therefore the impacts on LSF are expressed in kg C · years, or equivalent units. The meaning of this indicator is quite straightforward, although not evident: it refers to the amount of SOM which has not been present in the soil during a certain time, and thus it is connected to the life support functions this SOM has not been performing during all this time. Usually, other indicators in LCIA express the amount of a substance that causes impact (CO_2 for global warming, SO_2 for acidification, etc.); oppositely, the SOM indicator expresses the amount of a beneficial substance that is NOT present, and this is the reason of the impact.

IV.3.4. Consistency of SOM as an indicator for life support functions in the framework of LCIA

Apart from the scientific basis for using SOM as an indicator (section IV.3.1) and the practical considerations on data collection (IV.3.2) and procedure for using this indicator in LCA studies (IV.3.3), the consistency of the indicator within the framework presented in section IV.1.2 must be checked. The following paragraphs deal with the theoretical implications of SOM as an indicator of impacts on the life support functions, and with the consistency with the LCIA framework.

Consistency with the LCIA framework for land use impacts

The impact chain linking environmental interventions in the inventory to the relevant endpoints should be clearly described in order to assure a proper consistency of the indicator with the framework presented in section IV.1.2. The relationship of SOM with the Area of Protection Life Support Functions has been thoroughly explained in section IV.3.1. According to this relationship, loss of SOM is a negative impact, while an increase in SOM represents an increase in land quality. Therefore, SOM can be used as a land quality indicator in the framework presented in Figure IV-4. This assumption implies that substance cycles, soil fertility and climate regulation function better with high levels of SOM. In the case of water cycle, for instance, higher SOM levels imply higher water retention capacity, infiltration, and quality (thanks to the buffer capacity). Higher SOM levels also promote microbial activity, and thus the degradation and cycling of substances. It has been discussed that higher SOM levels are also associated to both chemical, physical and biological soil fertility. In the case of climate regulation, it may be argued that the current trend for carbon is the accumulation in the atmosphere, with the associated effect on climate through greenhouse effect; a higher accumulation in soil (as SOM) slows the process of atmospheric CO_2 building-up, and thus helps in climate regulation.

The mechanisms through which agricultural land use activities affect SOM level can be explained in terms of a carbon balance in soil (see Figure IV-12): if the inputs of carbon are higher than the outputs, then SOM builds up (quality rises); otherwise, it is being depleted (and quality decreases). Crop residues are the fraction of vegetable productivity not extracted by the farmer, i.e.: the free Net Primary Productivity (fNPP). In many systems, fNPP is the main source for reposition of SOM, and that is why it has been suggested that the inclusion of fNPP as an indicator for life support functions may be a rough approximation to SOM (see the discussion for Figure IV-9 in section IV.3.1, page 201). Another major source of soil carbon in agro-ecosystems is manure or any other organic waste added by the farmer. Nevertheless, the European Commission (COM 2002) alerts that specialisation and monoculture in herbaceous crops separates them from livestock production, and thus crop rotations with organic matter reposition are disappearing. The interventions that have to be described in the LCI phase are those agricultural practices affecting the SOM balance (Figure IV-12). Finally, the increase or decrease in SOM is the relevant environmental intervention to be obtained from the LCI through the application of SOM models and assessed by the LCIA.

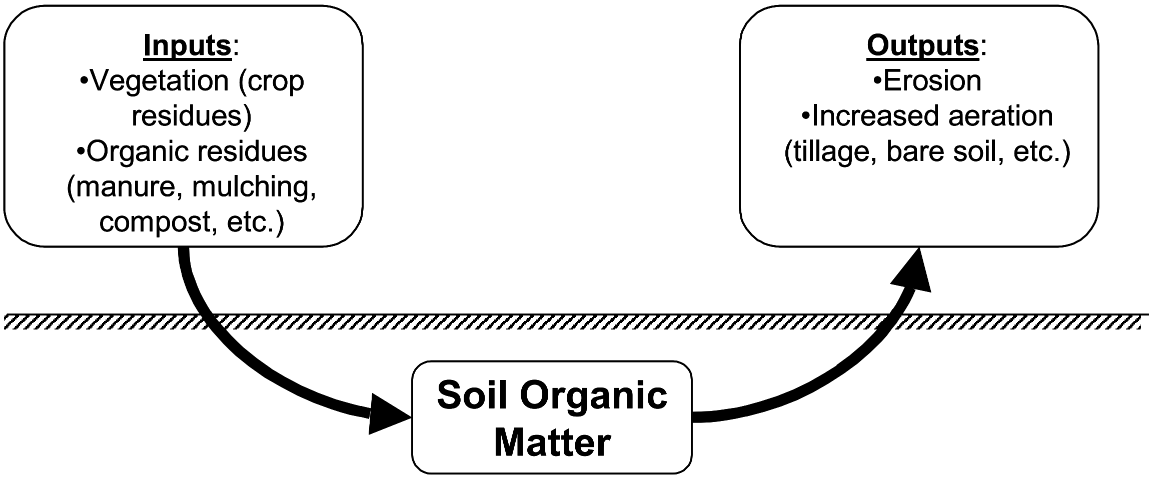

Figure IV-12: Items in the agricultural soil organic matter balance.

Threshold limit value

In the preceding paragraphs (see Figure IV-11) it has been stated that generally the soil quality loss in agricultural land use will not be so dramatic that soil will be unable to recover. However, if SOM is continually depleted with no reposition, soil structure may be damaged to a “point of no return” (or threshold or critical value; see Figure IV-13). At this point soil fertility is so seriously impaired that no biotic production can occur, and soil quality collapses even if land use stops. Then, organic matter inputs stop and soil further degrades until it is actually lost. Even though it is difficult to establish this critical value, Persson & Kirchman (1994) suggest that when Soil Organic Carbon (SOC) falls below 1%, soil structure is too damaged to continue biotic production. The European Commission also suggests this critical threshold to classify soils with less than 1.7% of SOM as in “phase of pre-desertification” (COM 2002). Arshad & Martin (2002) suggest using a value of 1% SOM or the average of the community (whichever is higher) as a threshold for sustainability. These values are not generally reached in croplands, though, as the residues left in the field (roots and other parts of the plant) are enough to keep the minimum amount of SOM.

It is thus suggested that if SOC falls below 1% (i.e.: SOM falls below ca. 1.72%) other degradation processes such as erosion must be occurring. If the threshold limit value is reached, the model suggested here would not be applicable.

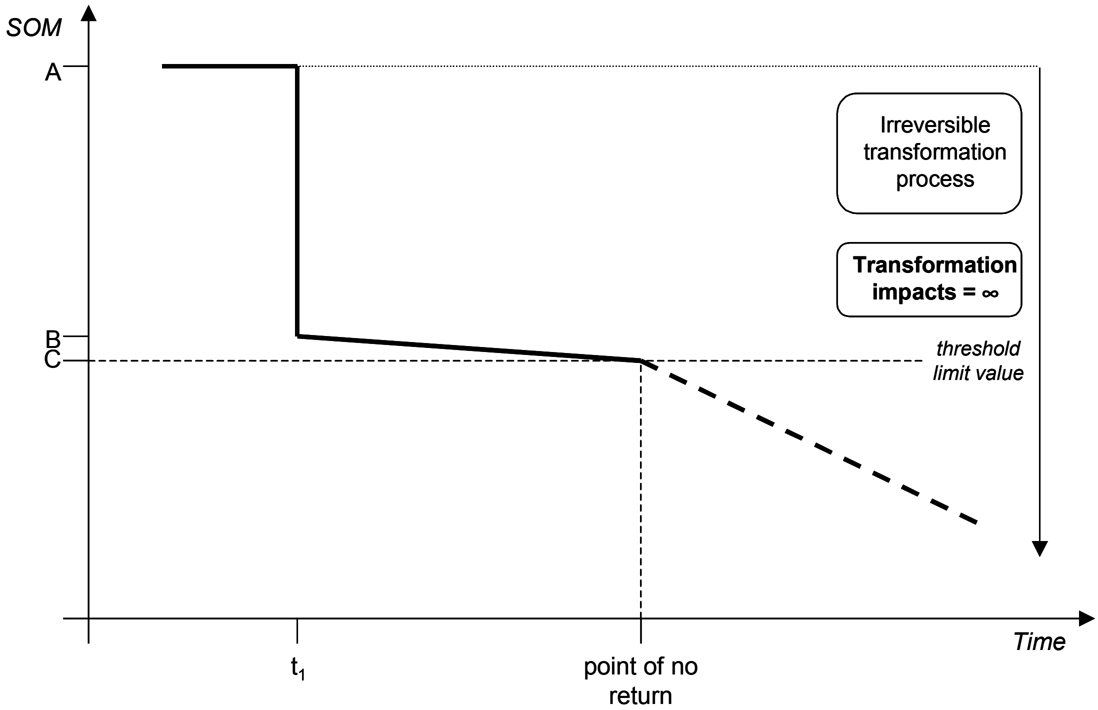

Figure IV-13: The concept of threshold or limit value.

The model is also limited at the superior level, because soil quality does not indefinitely increase with raising SOM levels. In the case of biotic productivity, Janzen *et al*. (1992) find that SOM has an effect on crop yield up to a level of 2% SOC[21]. Above this value, SOM does not exert any effect upon crop yield. Nevertheless, as SOM probably has a positive effect for other functions even above 2% SOC, it is assumed that any increase in SOM is positive. I.e.: soil quality is increasing when SOM is increasing. Besides, it may be added that many agricultural soils have SOM levels lying in the range of 1-2% SOC. Indeed, the European Commission states that 75% of soils in Southern Europe have SOC levels of less than 2%; In England and Wales, the proportion of soils with less than 2% SOC increased from 35% to 42% during 1980-1995, possibly due to a change in management practices (COM 2002).

Temporal scope

Changes in SOM levels due to agricultural practices are generally very slow. Apart from an initial dramatic decrease in SOM when forest or grassland is first converted into arable land, SOM decline due to further tillage and reduced organic matter inputs is a very slow process. Also when the management practices change and organic matter additions are increased, SOM slowly changes to a new steady state where oxidation equals inputs. Audsley *et al*. (1997, p. 57) report that these transitory states between equilibriums typically last around 20-50 years. This is actually one of the qualities that gives SOM its buffer capacity and its contribution to soil resilience. As a consequence, it can actually be stated that SOM is not a sensitive indicator to short-term impacts: soil degradation is not followed by a SOM decline in the short-term. Therefore, it should not be

[21] Considering that SOC is ca. 50-55% of SOM, this corresponds to a SOM level of around 4%.

used as an indicator for life support functions in short term LCIA, because it cannot show changes until the degradation or restoration process has been maintained for some years. Only LCA on the long-term effects of agriculture with time-scales of at least one or several crop rotations (see Cowell 1998, p.112), should thus use this kind of indicator.

Site-dependency

Section III.5 in the third chapter of this thesis offered a discussion on the effects of site-dependency on agricultural LCA results. It is suggested there that both physical site conditions (soil and weather) and producer's practices (technique) are crucial for the LCI results. Moreover, land use impacts need to be compared to a reference state, which depends on local potential, i.e.: they must be treated in a site-dependent way.

In relation to these findings, the model based on SOM as an indicator for impacts on life support functions considers soil and weather parameters ("physical site conditions", in Chapter III), as well as management practices ("technique") in its calculation procedure. Indeed, Table IV-7 shows different soil carbon levels for different soils of the world (dependency on soil type), and Table IV-8 (page 208) shows soil carbon levels in life regions (which are climate-dependent). Figure IV-12 highlights some of the typical technique-dependent aspects that affect SOM, and are consequently included by the SOM indicator. Besides, current level of SOM is compared to the local reference state (relaxation potential), as it is expressed in Equation 1 (page 211).

In summary, it may be concluded that the model for the assessment of impacts on life support functions of land based on the SOM indicator includes all the ingredients to consistently cover the site-dependency required by agricultural LCA. This is true both for the LCI results and for the impact assessment phase, where the system's effects on SOM evolution are compared to a reference state.

Scale

The indicator is working at the level of soil processes, where impacts on life support functions of soil are relevant. Therefore, the SOM indicator can be considered adequate for addressing the impacts on life support functions in agricultural LCIA.

Transferability

Today, there is ongoing work within the ISO for the standardisation of soil quality measures (Nortcliff 2002). This will notably increase the international acceptance of soil quality indicators, and the SOM indicator should include the conclusions of such standards. Mainly, this will possibly affect the methods used for the measurements, as well as the conditions of such measures: number of replicates, position within the plot, etc.

In the case of LCA, Blonk *et al*. (1997) suggest that the methods to include land use impacts in LCIA should be applicable to all ecosystem and land use types. The SOM indicator is possibly only advisable in the case of agricultural LCA, because its data requirements cannot possibly be fulfilled in industrial LCA. Besides, its sensitivity is not high for changes in the short term, which makes it undesirable for transforming types of land use (e.g.: if a piece of land is sealed because of a road construction, it may take some years to detect a significant reduction in SOM levels, while the effect is obvious for other indicators). Nevertheless, its ability for detecting even low levels of degradation at the soil level make it a very interesting indicator for LCA involving biotic production, such as agriculture or forestry. This is further discussed in the next section.

In practice, the SOM indicator will mainly be useful for comparing agricultural systems functioning under different conditions (technology, site, technique, etc.). Other LCA applications involving agricultural stages need further comment. A topical case is the comparison of products made of naturally grown materials with mineral- or petrol-based ones (e.g.: bio-diesel vs. diesel; leather vs. plastic shoes; wooden vs. metallic furniture; etc.). In this case, agriculture (or forestry) is to be compared to oil extraction or mining. The SOM indicator will be useful for the (reversible) occupation impacts on life support functions from agriculture or forestry, while oil extraction or mining generally exert an irreversible (transformation) impact on land. Therefore, the impacts involved, which have to be compared, are of a different nature, and not comparable at the midpoint level where SOM is evaluated. In other words, land use impacts from these land use types are different, and the preference of ones over the others remains a matter of valuation, which is not explored in this thesis.

IV.3.5. Evaluation of the suggested indicator

Some general items should be addressed when discussing methods for impact assessment (ISO 2000; Lindeijer *et al*. 2002). In the following paragraphs a thorough discussion is included on the method presented earlier in this chapter.

Representation

The indicator soil organic matter represents properly the vulnerability of life support functions of soil. Soil productivity is much affected by SOM due to its effects on soil physical, biological and chemical fertility. Besides, the fact that the relaxation time is also included in the procedure increases the environmental relevance of this indicator, because not only the degree of degradation but also the effects in the future are included. The good representation of soil quality by SOM is recognised by many authors, which gives international acceptance to the indicator.

Not all life support functions are consistently represented, though. Erosion protection is partially represented because high SOM levels reduce soil vulnerability to erosion by increasing water infiltration and aggregate stability. However erosion depends on many other factors, particularly on vegetation cover. The resource aspect of soil lost through erosion may be included in the abiotic resource depletion. To calculate the amount of soil lost, estimations of annual loss may be used if regional- or crop-specific data exist; otherwise, models such as the RUSLE may be used or even real measurements, if available, may be included in the analysis. Then, to characterise soil loss, methods such as the static reserve life are readily available (Cowell 1998). Otherwise, the application of other methods for characterisation may be studied. Particularly, it would be

interesting to explore the concept of the energy consumed for soil formation or regeneration (Blum 1997), because it seems to be parallel to the widely used EI99 (Goedkoop & Spriensma 1999).
Other aspects of land use impacts, chiefly related to the intrinsic values of land, need to be assessed separately:

- Biodiversity can be included using simplified indexes. Vascular plant species (Lindeijer *et al*. 1998; Müller-Wenk 1998; Köllner 2000) is one of the most promising indicators, though more sophisticated ones might be more relevant (Weidema & Lindeijer 2001). Biodiversity needs to be considered separately from life support functions because they represent different values of land: intrinsic and functional. Accordingly, a semi-arid area with a high biodiversity value due to the uniqueness of its species is no good from a life support point of view, and a change to a highly productive land would be seen as positive from a LSF point of view. E.g.: see maps presented by Lindeijer (2000b) to see that regions with the highest biodiversity values do not always match the areas with highest biotic productivity.
- Aesthetic value of landscapes should be included when relevant, using qualitative regional information. Nevertheless, its inclusion is questionable in studies that compare very different realities (e.g.: Sweden, Brazil and Malaysia in Mattsson *et al*. 1998, 2000), as very diverging values will be put on landscapes.

Level of sophistication

The use of SOM models in the implementation of this indicator allows for a full impact quantification, which requires for some mathematical complexity in the calculation procedure. In addition, the nature of these models assures a high degree of regionalisation due to the inclusion of climate and soil properties, along with agricultural practices, in the calculation.
SOM models are generally developed from empirical data and so the equations used in different models tend to give different results although most give similar trends. This is due to the fact that similar processes are modelled, but with slightly different rates due to local conditions. The tendencies given by SOM models rather than the actual SOM values are needed for the indicator.
Models should be validated before they are applied under new conditions. This represents a limitation because most models have been developed so far in Europe and the USA, and so they are not globally applicable. Besides, the uncertainty in the outcome of these models is high, due to the huge amount of parameters used. Nevertheless, the use of the method will be technically valid as long as the same model is applied using the same assumptions in order to compare different production systems. The uncertainties in predictions based on the use of SOM models should be taken into account when interpreting results (see section IV.4).
Obviously, the high site-dependency of the method requires for a case-by-case application, and so it limits the possibility of having standard characterisation factors. Nevertheless, this possibility remains a possible field of future research.

Sensitivity

Due to the fact that SOM evolution is very slow (which contributes to it its role in soil resilience), changes in its levels are hard to detect in the short term. That is, the effects of management practices on SOM level are only detectable when they have been going on for some years. This is the reason why the method based on SOM is not good in short-term studies nor possibly for transforming types of land use. Nevertheless, the time scope of the LCIA should not be mistaken for the time scope of the functional unit. Thus, a study could be based on the outcome of one year in one hectare, and still the impact assessment could use the whole crop rotation (or a series of crop rotations) in order to calculate the impacts due to the product system. In general, it can be said that the model suggested is relevant when comparing different agricultural systems, where time boundaries covering at least one crop rotation should be the rule. Indeed, one of the main advantages of the SOM indicator is that it allows for a sound comparison of maintaining types of agricultural land use from the point of view of impacts on life support functions.

Environmental mechanism

The main idea behind the method is that a decrease in SOM level represents a loss in soil quality and resilience, and therefore an impact on life support functions. The environmental mechanism that connects environmental interventions to losses in SOM level is complex, though fairly well understood. It is based on a soil carbon balance, which requires knowledge on inputs and outputs of organic matter to and from soil (see Figure IV-12). The effect of soil management practices on these inputs and outputs also needs to be known. Thus, the addition of organic wastes and plant residues contributes to SOM build-up, while reducing these inputs or increasing oxidation (e.g.: through tillage) contributes to SOM decline. The basics of the mechanism are easily understandable, and its complexities do not need to be fully understood thanks to the use of SOM models. In these models, the user must introduce site-specific data on vegetation type, cropping practices, soil and climate. Therefore, if the data used in the calculation is given in the LCA report, the results should be fully reproducible.

Value choices and assumptions

Some value choices and assumptions are implicit in the indicator based on SOM. This does not mean that the indicator is not valid, but it is necessary to make them transparent in order to be able to better understand the results.

- In the first place, it is noticeable that using SOM as an indicator in the way that has been suggested implies that higher productivity of the system (and thus higher SOM) is considered to be good. This observation might seem non-relevant, because if one is looking for an indicator for life support functions, obviously it is because these are important; hence, a life support function such as productivity should be regarded positively. Nevertheless, it must be clear that the life support functions of land are being protected with this indicator, and not others. Actually, there can exist ecosystems where productivity is naturally low, but rare species are found there and so it has a high intrinsic value. SOM (and any other indicator for life support functions) would conclude that the system is of low (functional) value. This assumption is related to what the indicator represents: SOM indicates the functional values of land, whereas intrinsic values must be analysed with biodiversity and other suitable indicators.

- Probably the most important assumption in the model presented for the impact assessment of life support functions is that a full recovery of the impacts always takes place. Therefore, no transformation impacts are considered, and only those related to the occupation process itself (occupation impacts) are attributed to the product system. Also the occupation impacts produced during the relaxation time that can be allocated to the product system are considered in the method suggested, and a consistent calculation procedure for this allocation is presented in section IV.3.6.
- The reference state chosen for the calculation of occupation impacts is the relaxation potential that can be reached within the scope of the study. It is suggested that the time scope should be long enough as to include the full recovery of the system to the original (natural) state. This assumption is considered to be consistent in agricultural LCA, where the life support functions of soil are not generally impaired irreversibly (i.e.: no transformation impacts occur). In this situation, only the current relaxation potential can be used as a reference state.
- In the event that the model predicts SOM level falling below a critical value (recommended at 1% SOC), the degradation might become irreversible. If this is the case, it is suggested that the use of the present method is no longer valid, and that new methods for land use impacts assessment including transformation impacts are considered. If the SOM indicator for life support functions is complemented with an analysis of erosion, a critical threshold value for the erosion could also be established on order to earlier predict the trend to irreversibility.
- During the calculation procedure, some expert judgement is required in order to determine the "most likely situation" in two scenarios: if the product system was not established and after the system is abandoned. The quality of the results can depend on these assumptions to an important extent, and so it is necessary that they are made clearly explicit in the report.
- Finally, it is important to bear in mind that the method suggested is highly dependent of SOM models. Therefore, the quality of the method used and the assumptions made in it are crucial for the results of the impact assessment. In general, methods that have been validated in conditions similar to the one in the product system (climate, soil and vegetation type…) should be used. When this is not possible, some validation using real measures should be done in order to check the model predictions with the actual evolution of SOM.

Consistency

Not all degradation processes represent a loss in SOM. Mainly erosion, toxic substances build-up and salinisation must be mentioned here. Of these, erosion actually does represent a loss of SOM, because the eroded soil is usually richer in SOM than the remaining soil[22]. Nevertheless, as SOM models do not usually include erosion effects on SOM, soil lost through erosion should be assessed as an impact on resource availability in the way suggested by Cowell 1998. Also the

[22] Pimentel *et al*. (1995) report that eroded soil usually contains up to 1.3-5 times more organic matter than the soil remaining.

build-up of toxic substances in soil has effects on soil quality that are not reflected by changes in SOM level. In this case, though, it can be assumed that toxic effects are sufficiently represented by toxicity impact categories. Finally, salinisation is a very specific problem for which no recommendation is done and which should be properly addressed in the cases when it can be relevant.

a proper allocation of relaxation periods can avoid double counting in successive land uses (as addressed in section IV.4; see Figure IV-18 in page 225). In order to do so, the relaxation process with and without the system under study must be estimated.

Using the relaxation potential as the reference state allows for a full consistency with the framework for land use impact assessment suggested by the SETAC working group on land use impact assessment (Lindeijer *et al.* 2002). The occupation impacts are thus fully represented.

Applicability

As already explained in the paragraph on "Transferability" of the indicator (see section IV.3.4) the SOM indicator is chiefly applicable to the comparison of agricultural or forestry systems. Land use impacts coming from life cycle stages different than agriculture are not properly represented by this indicator, and thus other indicators should preferably be used in those cases. This conclusion limits the applicability of the SOM indicator, but as LCA is increasingly applied to agricultural systems, its good representation of agricultural soil degradation processes still renders it a promising approach for impacts on life support functions.

The indicator can be applied using adequate SOM models for each system that has to be assessed, because it is extremely difficult to generate standard characterisation factors. Using estimates from standard figures such as the ones provided in Table IV-7 and Table IV-8 can do an approximation of the impact assessment. However, considering the high site-dependency of the method it is recommended that the SOM models should be applied case-by-case. In order to do so, some expert judgement, apart from data on the site characteristics, is needed for the modelling of the "most likely situation" during the relaxation process before and after the system. It is expected that such data and expert judgement can be provided in agricultural LCA. Section IV.4. Implementation of the indicator using SOM Models gives more details on the use of SOM models.

IV.3.6. Allocation issues

In agricultural LCA, it will generally be the case that different activities (product systems) will take place over one piece of land in succession, thus causing varied impacts on land. A fair distribution of all the impacts amongst the different product systems should be performed. A straightforward way of allocating the impacts is charging the system under study for the impacts of which it is immediately responsible, as shown in Figure IV-14.

Reference case

In Figure IV-14, a certain type of land use is shown that could describe the general situation in most European crop fields. A natural forest was chopped down many years ago (t_1) with the subsequent decline in SOM. With agriculture, SOM continued declining but at a much slower rate. Eventually, agriculture will be abandoned and the field will be invaded by forest (t_2), rising SOM levels probably back to the original state (A). We want to study the impacts caused by a certain type of agriculture during a limited lapse of time comprising a couple of rotations (from t_{ini} to t_{fin}). We know the SOM level before our system started (B) and just after the field left the studied system (C). Therefore, knowing the relaxation potential (which, assuming no transformation impacts, will be equal to the original state A, probably that of a natural forest), we can readily calculate the occupation impacts due to the land use. The area with vertical dashed lines represents this. But soil quality is not the same when soil leaves the product system, and this has to be taken into account. The way of doing so is to estimate the lapse of time it takes the re-naturalisation process to reach the SOM level prior to our system (B). This extended time (relaxation time, from t_{fin} to t'_{fin}) is to be added to the system under study, and hence the dotted area (occupation impacts during re-naturalisation) is allocated to it.

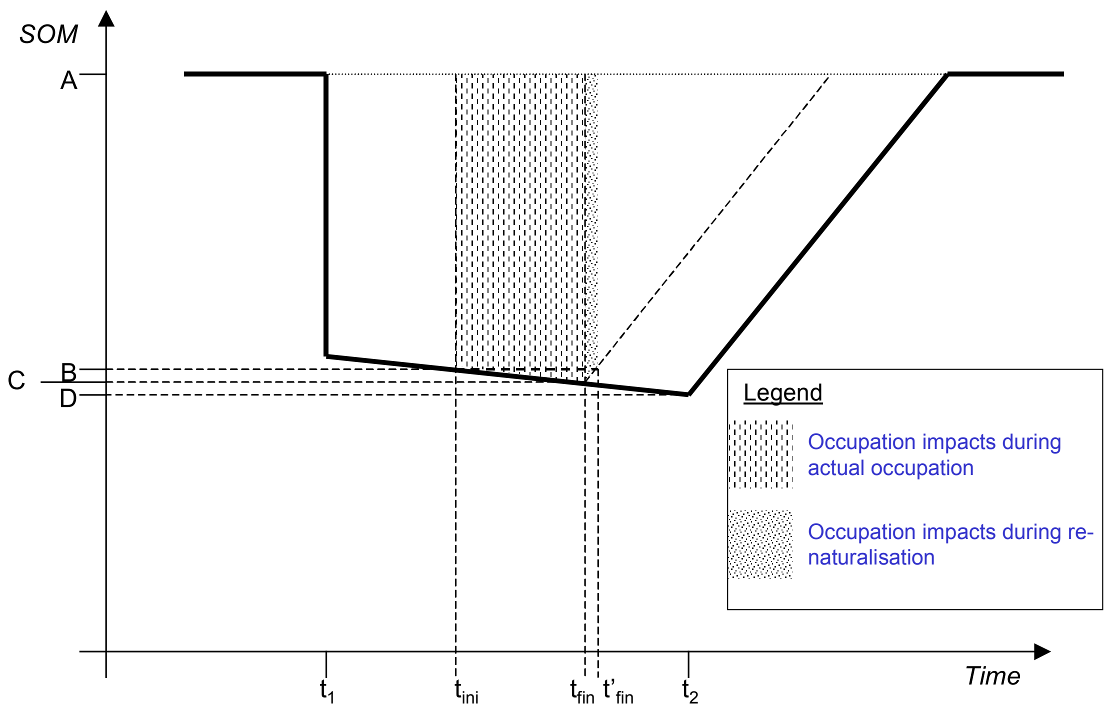

Figure IV-14: Allocation of impacts on life support functions.

Note that the relaxation process in Figure IV-14 is not actually taking place until many years later (at t_2), because other crops are produced in the same field after the crop under study, further reducing SOM level and retarding the re-naturalisation (bold line). Nevertheless, this is no reason why one can not use this hypothetical advanced recovery time with the purpose of estimating the effects of the studied system.

Case 2: Reduction of the recovery rate

In Figure IV-14 it is considered that the rate of relaxation (slope of the recovery line) is not changed by the land use. It is more likely that bigger impacts will be harder to recover. I.e.: at quality C (when the soil "leaves" the product system, t_{fin}) the recovery may be slower than at quality B (when the soil "enters" the product system, t_{ini}), because the soil starts from a worse position (lower SOM level). Therefore, it can be argued that the system under study has not only delayed the occurrence of the re-naturalisation, but it has also increased the relaxation time (and thus it follows relaxation curve *b* instead of *a* as shown in Figure IV-15). Consequently, the additional occupation impacts due to the extended relaxation time are not only those in the dotted area between t_{fin} and t'_{fin}, but also the dotted area between curves *a'* (same slope as *a*) and *b*.

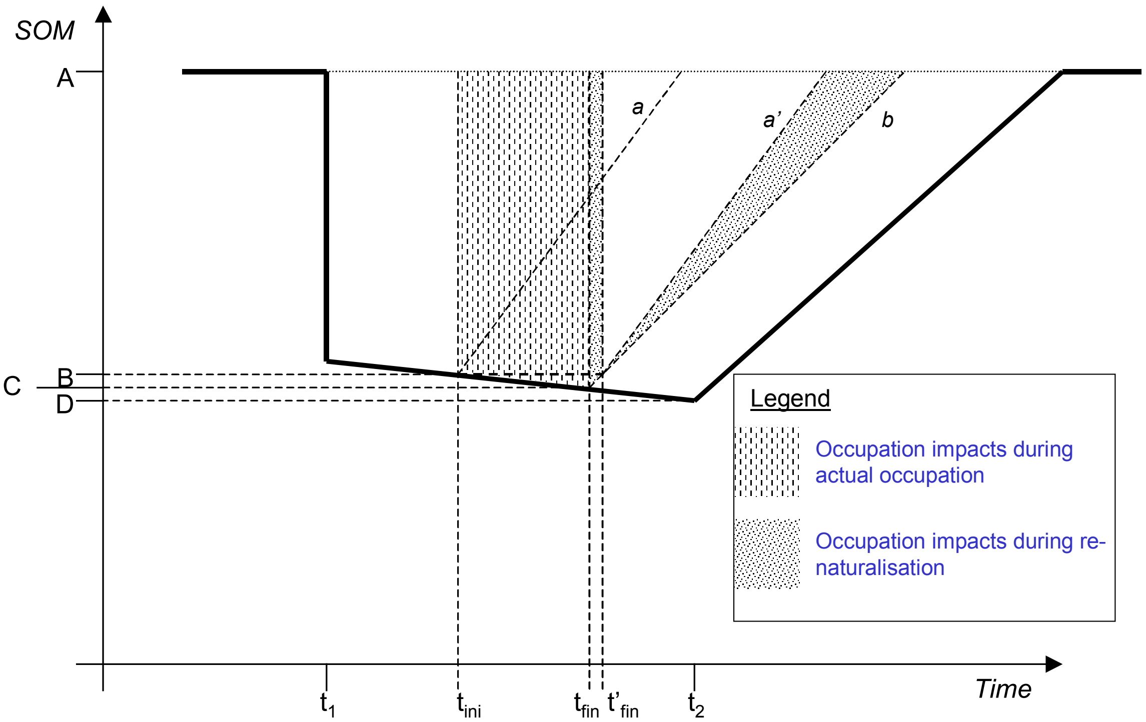

Figure IV-15: Allocation when the relaxation rate is affected by the system under study.

Case 3: Land use increases SOM

In contrast with the situation presented in the former figures, Buyanovsky & Wagner (1998) suggest that modern agriculture is in many circumstances increasing the levels of SOM, mainly due to increased productivity that leaves higher amounts of organic wastes (roots and plant residues) in soil. This situation raises still another case of allocation, as the system may be actually improving soil quality (Figure IV-16).

In the case shown in Figure IV-16 the system leaves the soil in a better quality (level C in SOM) than it found (B). This case is probably common in the real world in fields that were cleared from forests or natural grasslands (quality A) and sustained agriculture that did not counterbalance carbon losses. Then, at point t_{ini} agricultural practices changed to a better management of SOM so that SOM can be recovered. Eventually, the field is abandoned (t_2) and a natural ecosystem that recovers original SOM levels is established. Figure IV-16 considers that the relaxation process is still a bit slower than it would have been if the soil had been left undisturbed when the system began (t_{ini}). This fact is included by the dotted area, which represents the occupation impacts allocated to the system due to the retardation of relaxation. Actually, if the system does not change the relaxation rate (slope of curve "a"), then this extended occupation impacts would be zero. It is important to note that even though SOM is rising during land use, the system is still causing occupation impacts on life support functions, because it is maintaining soil in a quality lower than its potential.

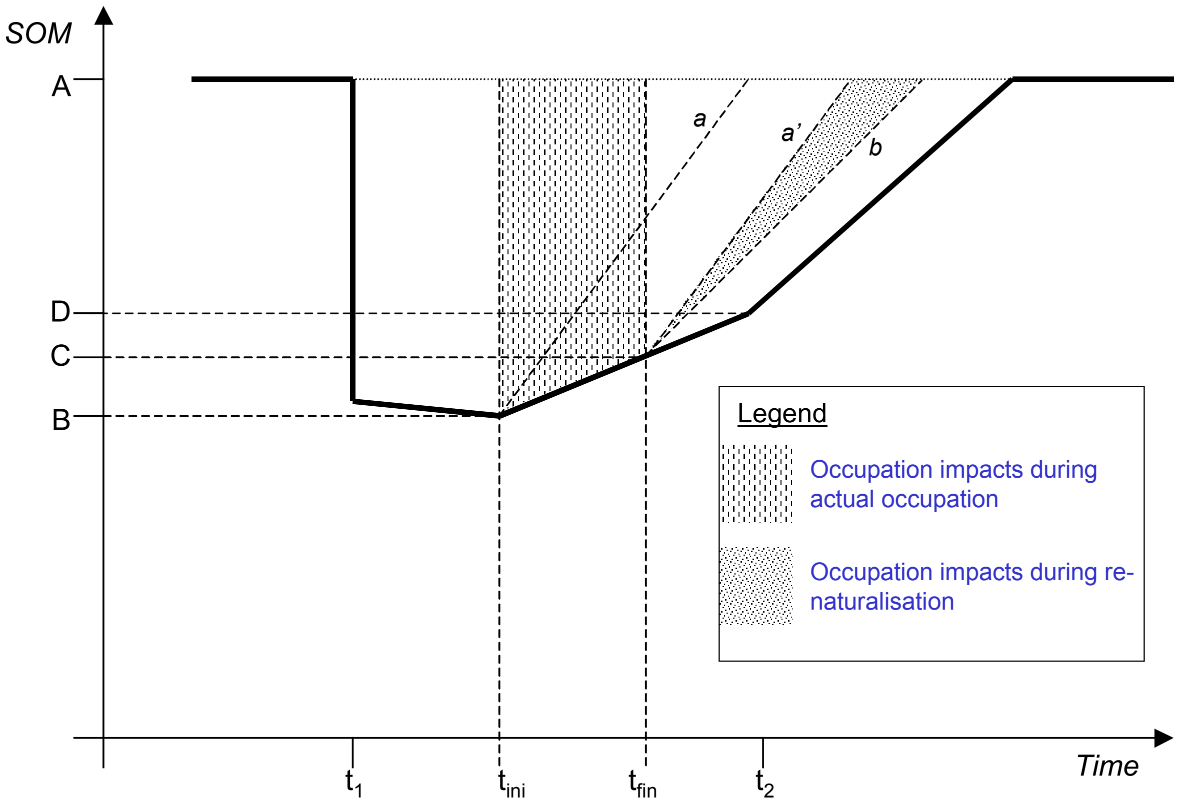

Figure IV-16: Allocation of occupation impacts in a system that enhances soil quality.

An extreme case might be described when the system does not only increase soil quality, but does so at a rate faster than the natural relaxation (e.g.: "b" is steeper than "a" in Figure IV-16). Here, the final relaxation time would not be extended, but actually reduced, and so the system under study should be credited for it. Again, the way of dealing with this is to calculate the impacts that would have been produced if the soil was left undisturbed before the system began (line "a"), and subtract this value from the impacts produced by the system relaxation (line "b").

Case 4: Allocation in successive land uses

All the situations presented until now can be easily understood separately. Nevertheless, when considering a series of several product systems some double counting problems may arise when trying to allocate the occupation impacts caused during relaxation. This situation is depicted in Figure IV-17, where 5 different systems (including system 0 –S0-, the initial transformation) are using the same piece of land consecutively. In the situation shown in the figure, the initial product system (S0) extracts all wood from a forest leading to a SOM depletion. Then, successive agricultural systems (S1 from t_0 to t_1; S2 from t_1 to t_2; S3 from t_2 to t_3; S4 from t_3 to t_4) are established until eventually land is abandoned in t_4 with the consequent start of re-naturalisation. Occupation impacts are shown for each of the product systems, including the impacts from the actual occupation plus the impacts during soil relaxation.

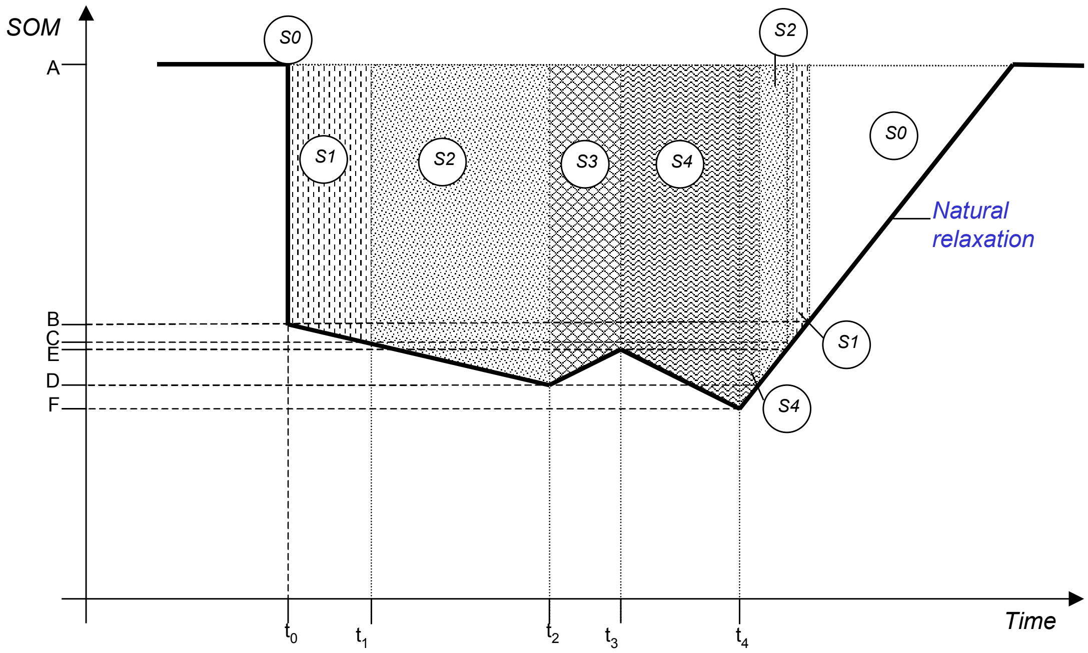

Figure IV-17: Allocation problems in successive product systems.

As system 3 rises soil quality it is not quite clear which occupation impacts should be allocated to the systems immediately before and after this system (S2 and S4). Also, from Figure IV-17 it seems clear that S3 should not be attributed any impacts during relaxation, even though the relaxation rate in S3 is slower than the natural relaxation rate (see Figure IV-16).

All these difficulties can be numerically solved by integration: approximating the areas on each system's relaxation curves by multiplying SOM levels at each time interval (1 month- or 1 year…) and adding them up. Then, the area under the relaxation curve before the system (e.g.: t_{ini}) is subtracted from that starting at the end of the land use (e.g.: t_{fin}) (see Figure IV-18).
In Figure IV-18 each system is allocated the impacts directly attributable to it. Hence, the relaxation process that would occur if the system did not start is fully attributed to the previous system. Obviously, in reality this relaxation process would start many years later (for instance, the relaxation for the disturbance at t_0 shown in Figure IV-17 does not occur until some time after t_4, when quality B is reached). Nevertheless, the graphical calculation shown in Figure IV-18 avoids any allocation problems and double counting, and can easily cope with changes in relaxation rates such as those shown in Figure IV-15 and Figure IV-16.

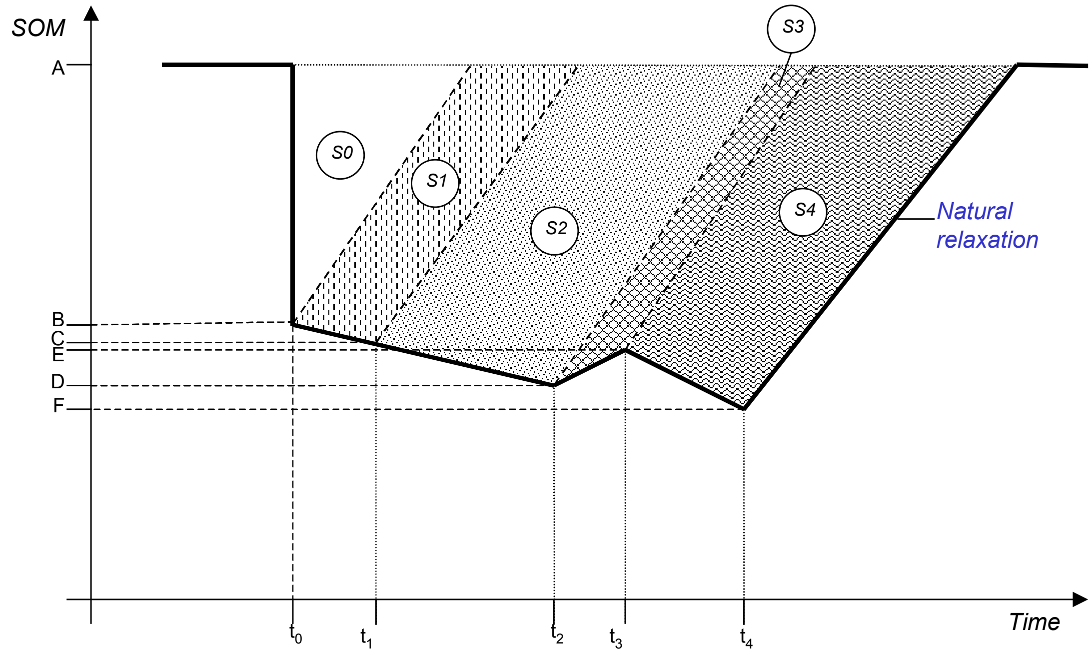

Figure IV-18: Allocation using the impacts directly caused by each system.

IV.3.7. Consequences of the carbon cycle modelling for global warming impact assessment

Even though the effects on climate were not a main concern in this thesis, the tendency found in SOM evolution for a specific system should be taken into account when expressing the agricultural stage's contribution to global warming. This point is already raised in Chapter II (subheading "Allocation" in section II.3.2.) and in Chapter III (in "Life Cycle Impact Assessment", section III.1.2.). It has also been an iterative matter of discussion in many references (see e.g.: Guinée *et al.* 2002). It is often suggested that CO_2 fixation by plants should be considered as a "negative emission" (with beneficial effects) of the agricultural stage. Subsequent CO_2 emissions associated to vegetable materials should thus be considered as emissions, even though that CO_2 is considered to be "renewable".

Total CO_2 fixation by plants' photosynthesis, though, is difficult to assess, and it is usually approached from the carbon content in the harvested crop. This value overlooks the amount of carbon fixed by roots and other plant residues that are left on the field. In addition, soil carbon emissions occurring when SOM is depleted are never taken into account, even though these can be highly relevant, as it has been pointed out above (see the discussion for Figure IV-10, page 206). It is here suggested, thus, that only SOM built-up should be considered as a negative emission from agriculture, while SOM decreases detected with the application of the SOM indicator should be considered as CO_2 net emissions from agriculture. Subsequent carbon emissions from agricultural products may be disregarded because they come from short-term carbon cycle. Some authors suggest including carbon emissions from these products when they are in a form different than CO_2 (e.g.: CH_4).

IV.4. Implementation of the indicator using SOM Models

Section IV.3.5 concludes that models describing SOM evolution with time are required in order to calculate the relaxation time and potential. This section starts with a description of SOM models and its main characteristics, including the main processes that rule the carbon cycle. The procedure for calculating the impacts on life support functions is then explained, together with a practical example, in section IV.4.2.

The implementation of the SOM indicator according to the framework for land use impacts (presented in section IV.3.3) requires that some parameters are known:

- The level of SOM during the occupation process,
- The relaxation potential that can be reached at any moment during the land use (Reference State),
- The time it would take SOM to reach the relaxation potential if the land use was ceased (relaxation time).

As stated in section IV.3.2, some of these figures can be obtained using real measures or expert estimates, together with literature data (see Table IV-7 and Table IV-8). Nevertheless, estimates of reasonable quality are very hard to get, particularly for the relaxation time (which not only depends on soil and climate, but also on more site-specific conditions, such as land management). In order to calculate these figures, models can be used that describe the evolution of SOM in time.

IV.4.1. Introduction to SOM Models

A model is an abstract and simplified representation of reality. SOM models in particular are mathematical functions that represent the dynamics of carbon (and usually also nitrogen) in soil-plant systems (McGill 1996). They are usually mechanistic (functional) models, in the sense that they try to describe all the relevant processes affecting SOM in order to predict its evolution. Consequently, a thorough knowledge of the carbon cycle is needed (see Figure IV-19). Generally, SOM models use first order kinetics to predict SOM synthesis and degradation, with different rates of degradation in the different pools, and material flows between these pools are estimated. Once these processes are described, they must be calibrated using real SOM measures: degradation and transfer between the different SOM pools rates are generally obtained from long-term SOM datasets. As a consequence, the results of the model are generally dependent on the conditions of the site where the model was developed, which introduce a high degree of site-dependency in these models.

The carbon cycle
Figure IV-19 shows a simplified representation of the carbon cycle processes commonly included in SOM models. The structure of the models varies widely: some have sub-models for plant

growth, soil moisture and nutrients separated from the SOM evolution, while others focus chiefly on SOM. Nevertheless, they usually try to model the processes shown in the figure by introducing the rates of transfer between pools.

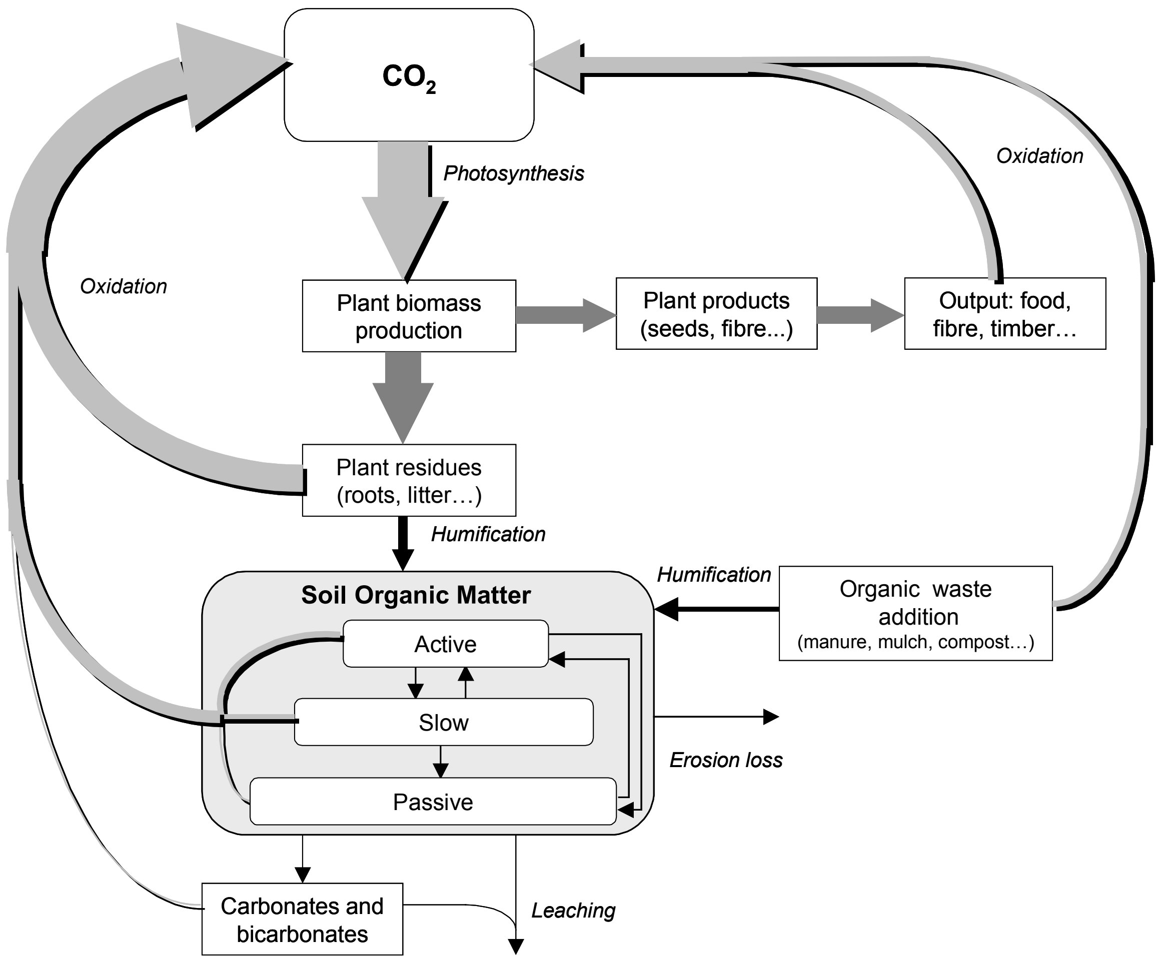

Figure IV-19: Simplified Carbon cycle in an agro-ecosystem (adapted from Brady & Weil 1999, p. 471). The thickness of flows shows their likely relative importance.

As shown in the figure, Soil Organic Matter is mainly formed from the *humification*[23] of organic wastes (plant residues, plant roots, manure and organic wastes added to soil) entering the soil. Also synthetic processes inside the SOM pool tend to polymerise and form complex substances, resulting in the differentiation of different pools: *active*, *slow* and *passive* SOM[24]. Loss of SOM is

[23] Humification is the process in which chemical reactions, biological and microbial processes and synthesis, transform organic matter into high and low molecular weight compounds that are not found in living matter. Generally, the process of humification increases the chemical complexity of compounds, making them more resistant to degradation (Porta *et al*. 1994).

[24] These pools receive different names in different models and references (Christensen 1996). Accordingly, the active organic matter is also named labile, decomposable or metabolic organic matter; slow organic matter is also called resistant or structural; and the passive organic matter, corresponding to the most complex substances, is often referred to as inert, recalcitrant, or simply lignin (even though this is not the only substance in this SOM pool).

mainly due to *oxidation* (particularly in arable agro-ecosystems), which affects primarily the active pool (formed by low molecular weight compounds, readily degradable by soil organisms). Erosion is another important path of SOM loss, albeit it might be around one order of magnitude smaller than oxidation (Brady & Weil 1999, p. 471). Leaching and incorporation of carbon into carbonate rocks are minor output paths of SOM. Lal (1998b) thoroughly reviews the processes leading to carbon emissions from soil (i.e.: SOM loss), including their relative importance, and he adds loss by methanisation as a relevant process in anaerobic soils.
Agricultural management clearly affects SOM by determining the carbon flows shown in Figure IV-19 (see Figure IV-12, page 213). As for the input flows, the farmer directly decides the amount and types of plant residues left on the field and organic wastes applied to soil, while plant roots are a more or less stable input (roots are seldom retired from soil). Also SOM outputs are affected by agricultural practices: the type of tillage directly influences soil aeration, and thus SOM oxidation, and erosion is heavily determined by tillage and other practices.
Site conditions are also important for SOM evolution, and these are chiefly incorporated in SOM models through soil parameters (e.g.: clay content and moisture) and climate data (e.g.: rainfall and temperature). The model user may generally introduce these data, so local conditions can easily be suited in the calculation procedure. Some models even allow for the incorporation of CO_2 concentrations in air, in order to check its effects on SOM evolution.

Powlson *et al*. (1996) offer a thorough review of SOM models based on long-term datasets, and a summary of the main characteristics is given for some of them in Table IV-9. Other methods may be found in the literature (e.g.: Petersen *et al*. 2002).

Table IV-9: Some of the main SOM models based on long-term datasets.

Model name	***Main references***	***Origin of data***	***Comments***
CANDY	Franko *et al*. 1995	Bad Lauschstädt, Germany	C & N dynamics Daily time step
CENTURY	Parton *et al*. 1987; 1988	Colorado, USA. Tested with data from Pendleton (Oregon, USA), Sweden, and Australia	C, N, P & S dynamics Grassland, agricultural, forest, and savannah systems Monthly time step
DAISY	Hansen *et al*. 1990; 1991	Denmark Validated under various conditions: Denmark, Germany	C & N dynamics 1 hour time step for soil processes; daily time step for crop submodel
ROTHC	Jenkinson *et al*. 1987	Rothamsted Experimental Station (UK); arable soils in a temperate zone.	C dynamics Monthly time step
Verberne	Verberne *et al*. 1990	The Netherlands	Submodels for SOM, inorganic N, crop growth, and soil moisture

The conditions (mainly weather and soil types) in which a model was created should be similar to the conditions of the study where the SOM model is to be used. Otherwise, a consistent validation with real data should be performed, in order to check that the model correctly predicts the SOM evolution in the new conditions. Nevertheless, it must be kept in mind that process-based, mechanistic models are not designed for the accurate prediction of ecosystem evolution, but mainly to understand and explain the tendencies in this evolution due to changing environmental conditions. Besides, these models cannot be tested in a rigorous statistically sound sense (Mohren & Burkhart 1994), although the general understanding of ecosystem processes is enough for the purposes of their use.

Input data for SOM models

One of the most commonly cited drawbacks of mechanistic models is their over-parameterisation, or huge need for real data in order to function. Typical input parameters include:

- Climate (data on air temperature and precipitation are generally enough; some models allow for the introduction of long-term weather data sets in order to estimate yearly variations in climate effects),
- Soil properties (texture is one of the most relevant attributes; of course, initial SOM level is required in order to start the calculation),
- Soil management (effects of fertilisation, tillage, organic matter additions, etc. are included by most models, as they are developed to study the carbon cycle in agro-ecosystems),
- Vegetation type (mainly the characteristics of vegetable organic matter are needed, in order to assess for the degradability and allocation to the different SOM pools).

As mentioned before, also the rates of transfer between carbon pools are needed for the model to run. These rates may vary between different ecosystems, and thus if no specific data are available some validation of the model is required before it can be used.

These requirements may result in long lists of data, which could hamper the applicability of the models. Nevertheless, most of the data required can be provided by the model itself for a number of possible situations, or easily estimated by the agricultural LCA practitioner with the help of agronomists.

With these data, the models can already be used in order to calculate SOM evolution during the land use.

IV.4.2. Calculation procedure and example of application of SOM models

This section explains the steps for the calculation of impacts on life support functions of land based on the SOM indicator suggested in section IV.3. The use of SOM models in its implementation is also illustrated with the application given by Romanyà *et al*. (2000). The aim of this application was not to assess soil quality or the effects of agricultural land use on life support functions, but to predict carbon sequestration in Mediterranean forest soils. Nevertheless, it may still be used for our purpose, because the calculation procedure is the same, and a hypothetical LCA study may be fitted perfectly in it. In the following paragraphs, thus, brief theoretical explanations are supported with how each step would have been performed in the example.

History of the land use

To start running the model, a steady evolution of SOM before the land use is generally calculated. The historical most likely events and land uses are introduced in the model, so that a picture of what has historically occurred until the SOM level prior to the land use was reached can already be gained. Then, the practices during the agricultural land use should be reproduced in as much detail as possible, so the evolution of SOM during the agricultural land use can be predicted. Finally, the land use ceases and the evolution of SOM must be assessed until the relaxation potential is reached.

Romanyà and colleagues assessed SOM evolution in old cereal fields that had been abandoned by the half of the XXth century. Pines were planted after the agricultural use, and have been growing since then. As fields afforested in different years were present in the area under study, SOM measures were available that could be used in order to validate the predictions of the two SOM models used: RothC and Century. In brief, the history of the place is:

- In 1750, the original forest (holm oak, *Quercus ilex* L.) is chopped down and cereal crop fields are established.
- Cereals were grown during ca. 200 years, with periodic addition of manure and crop residues, as well as a fallow year every 10 years.
- Around 1950 agriculture was abandoned, and a pine (*Pinus radiata*) plantation was established.

SOM evolution and calculation of the steady state

As defined in section IV.1.2 (see Figure IV-4 in page 179), the relaxation potential is the steady state that can be reached by re-naturalisation after a degradation process has taken place. In order to calculate it using SOM models, the situation most likely to occur after the land use must be assessed. Generally, this situation will be that of a natural recovery process, where different vegetation communities will appear on the land following a natural succession. Then, the advice of ecologists would be needed in order to establish the most likely succession in that site. Otherwise, human-induced vegetation could succeed if rehabilitation practices are used after the land use. In this case, the situation could be determined more easily.

Whatever is the case, the information on the different types of vegetation that are successively established in the land and on the land management (if any) should be introduced in the SOM model in order to calculate SOM evolution. Note that only the data possibly changed by the land use (i.e.: type of vegetation and land management) need to be filled in the models, as data on climate and soil should remain more or less unchanged.

The model should then be run until a steady state is reached in the temporal scope of the analysis, which will probably be similar to the state prior degradation. The time lapse until the steady state is reached is the relaxation time, and the SOM level in the steady state is the relaxation potential that should be used in order to calculate the occupation impacts.

For modelling purposes, the authors first run the models for 2000 years with a mature holm oak forest (the most likely vegetation) and a scheduled fire every 300 years, in order to obtain the initial

SOM level (3,075 gCm^{-2}). Local weather conditions were used. Then, the most likely agricultural practices were introduced into the models and they were run for 200 additional years with these conditions. Finally, SOM evolution under the *Pinus radiata* plantation was assessed. Figure IV-20 shows the SOM level evolution during all these years.

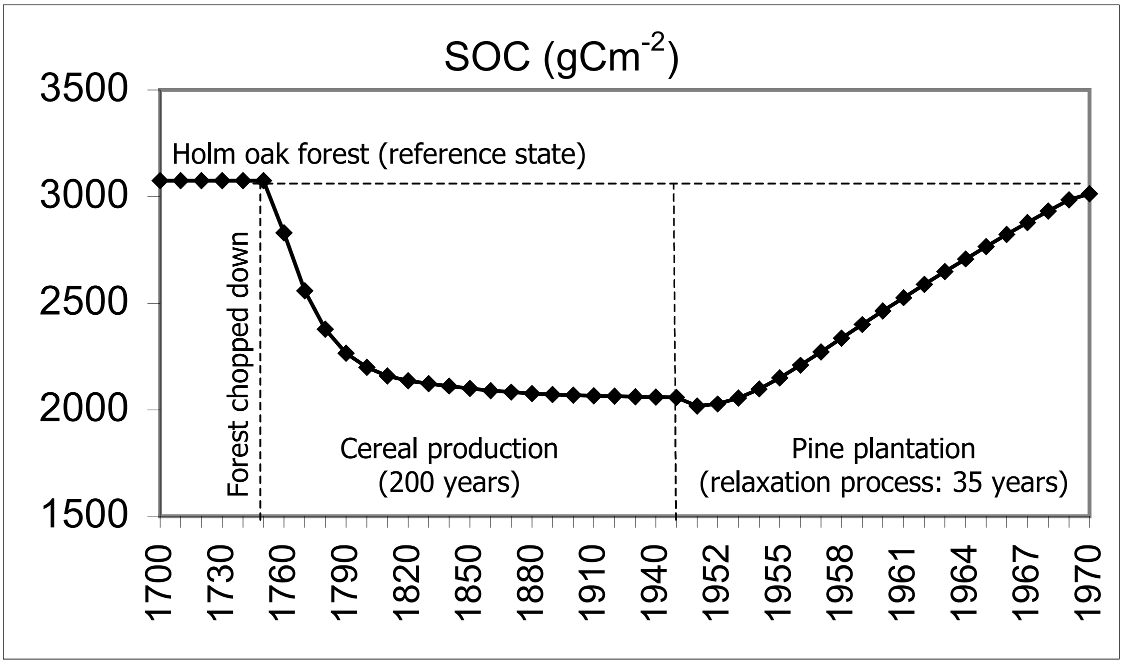

Figure IV-20: SOC evolution predicted by the Century model (Romanyà *et al*. 2000; data kindly provided by Dr Joan Romanyà).

Each point in the figure is the estimate of SOC for each of the years of the simulation. Note that the time step is 10 years until 1950; from 1950, a more detailed time step (1 year) is depicted. The relaxation process is thus very fast (ca. 35 years), and the slope of the curve should be much more steep. This quick recovery is possible due to the fact that is human-induced: the pines are artificially planted, and there is no need to wait for the natural recovery process. It is interesting to note that SOC levels gradually decrease under the cereal cropping, even though organic matter is periodically added in the form of manure (100 g C m^{-2} $year^{-1}$) and crop residues, and that there is a fallow year every 10 years. After agriculture is abandoned and pines are planted, SOC still decreases for a period of two or three years, while pines are still not producing enough litter to counterbalance oxidation.

Calculation of the land use impacts for LCIA

The predicted relaxation period (35 years) corresponds to the recovery of 200 years of degradation up to the relaxation potential, which is set at the natural forest level. Romanyà *et al*. (2000) actually point out that the steady state reached with the pine plantation is higher than the natural one (3,461 g C m^{-2}). In spite of this, the reference state has been kept at the natural forest level for this example. In a real LCA situation, impacts on life support functions might even be subtracted as

"negative emissions" (positive effects), bearing in mind that possibly a trade-off would have occurred with other impact categories (e.g.: reduced biodiversity quality of the pine plantation compared to the natural holm oak forest).
For LCA purposes, the functional unit will not be generally a cropping period as long as 200 years, but shorter periods (such as the 10 years of a crop rotation) or the production of a certain amount of product (e.g.: 1,000 kg of grain). As an example, the cropping period 1930-1940 is assessed, and impacts need to be described per tonne of grain (assuming a yield of 2,000 kg grain ha^{-1} during the 200 years of cereal production). Thence, an allocation problem arises. In this case, the general procedure described in Figure IV-18 cannot be used because the re-naturalisation process is not known and cannot be fitted in the data provided by Romanyà *et al.* (2000). Instead, and having in mind that SOM evolution is not complex because the same land use has been maintained during the years, the procedure suggested in Figure IV-17 will be used. It is assumed that none of the years of cereal production is having a bigger influence on SOM recovery, and thus the recovery time is equally allocated to the 200 years of cereal production. This is of course a simplification, because the first years would naturally be depleting more SOM; indeed, the first dramatic decline in SOM might be attributed to the wood extracted from the holm oak forest, rather than to agriculture. Nevertheless, to keep the example as simple as possible, an equal allocation is chosen. With this hypothesis, the total land occupation of agriculture during 1930-1940 is:

Actual occupation:

$$L_{a,actual} = 1ha \cdot 1year = 10ha \cdot year$$

Occupation during recovery:

$$L_{a,recovery} = 1ha \cdot \left(\frac{35year}{200year} \right) \cdot 10year = 1.75ha \cdot year$$

Thus, total land occupation is:

$$L_a = L_{a,actual} + L_{a,recovery} = 11.75ha \cdot year$$

In order to calculate the occupation impacts, Equation 1 (page 211) is used. As the function followed by SOM is not exactly known, an approximation with the values per each time interval is used. Then, the average SOC value between 1930 and 1940 is calculated and multiplied by the number of years (10) and the reference area (e.g.: 1 ha) to get the occupation impacts in g C·year (see Table IV-10). This is represented in Figure IV-21, where the occupation impacts during the actual land use and the occupation impacts during recovery are expressed. Note again that the recovery time is represented with a shorter time step, and thus, even though the area seems to be bigger, the impact is actually smaller.

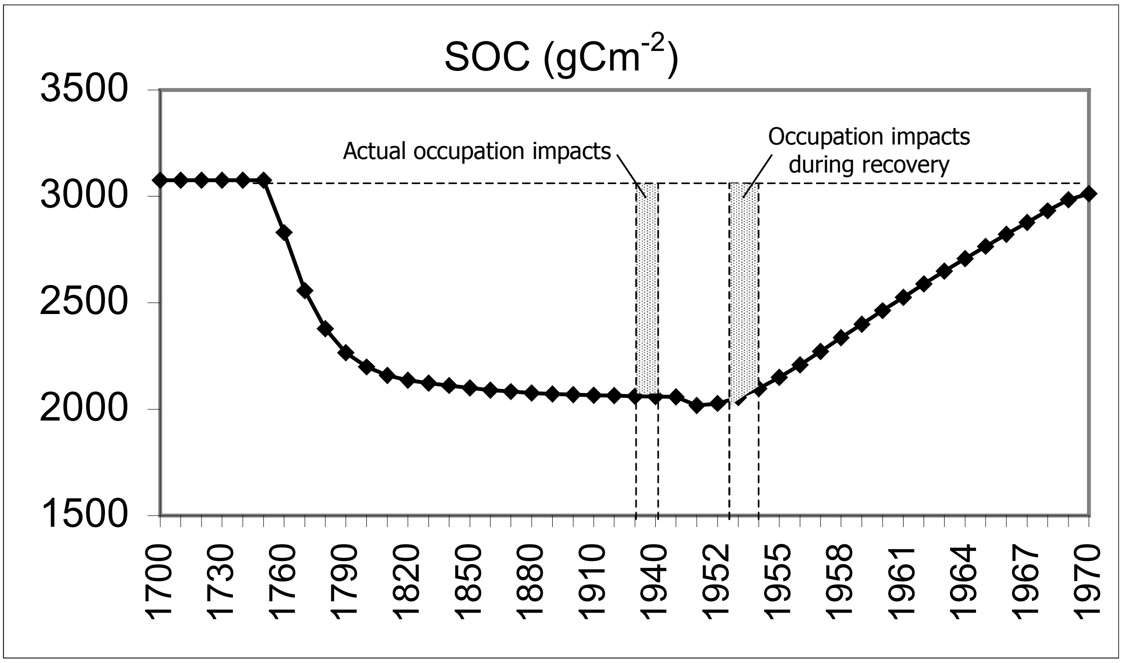

Figure IV-21: Occupation impacts of the agricultural land use between 1930-1940.

If the results are to be expressed per another functional unit, such as 1 tonne of grain, yield needs to be known (2,000 kg ha^{-1} $year^{-1}$). Then, the results expressed per ha may be modified to express them per tonne, taking into account that there is a fallow year every 10 years (see Table IV-10).

Table IV-10: Occupation impacts on Life Support Functions expressed per 1 ha and per 1,000 kg of grain, produced during 1930-1940.

Reference (f.u.)	*Occupation impacts 1930-1940*		
1 ha	Actual	101,543	kg C·year/f.u.
1 ha	Recovery	4,781	kg C·year/f.u.
1 ha	**TOTAL**	**106,324**	**kg C·year/f.u.**
1,000 kg	Actual	5,641	kg C·year/f.u.
1,000 kg	Recovery	266	kg C·year/f.u.
1,000 kg	**TOTAL**	**5,907**	**kg C·year/f.u.**

Note that the recovery time would have probably been longer if natural succession had taken place instead of the afforestation process. In the example, the recovery process was perfectly known because it has actually taken place, which is not very common. If the approach to calculate occupation impacts had been taken back in 1940, possibly a natural succession would have been assumed, and bigger occupation impacts would have been estimated. Then, the pine plantation taking place years after would have been considered as another type of land use, with negative land use impacts (that is, beneficial effects), because it would be actually shortening the recovery process. To be consistent, thus, if the afforestation process is considered to be not a new land use but a restoration activity, the impacts associated with it (chiefly energy consumption for producing and planting the trees) should be allocated to the cereal production. A typical case of trade-off between land use impacts and other resources consumption would be occurring.

IV.5. Conclusions for land use impact assessment in agricultural LCA

Some of the main conclusions that can be drawn from the consideration of land use in LCA are summarised below:

- Land use impacts from agriculture are highly relevant at a global scale. Besides, these impacts are an important difference between types of agricultural technologies, and should therefore be properly assessed by LCA when comparing such technologies. Comparing farming systems requires methodologies that allow for the assessment of maintaining types of land use.
- In the case of impacts on biodiversity, a proper framework for land use impact assessment already exists in LCA, both for transforming and maintaining types of land use. However, impacts on the life support functions of land still lack a consistent methodology.
- Some methods suggested so far for the LCIA of life support functions are based on productivity (NPP and fNPP), which indicates the short-term functionality of a system, but not soil quality on the long-term. Besides, due to their low sensitivity, productivity-based indicators cannot be used in a consistent way for the comparison of agricultural systems (and, in general, of maintaining types of land use). Other indicators have been suggested that are more closely correlated to soil quality than productivity, but these are usually less consistent with the LCIA framework for land use impacts.
- Intensive types of agriculture may produce higher yields than less intensive ones, usually at the expense of degrading soil quality (e.g.: depleting soil nutrients and organic matter, leading to compaction and loss of structure…). This reduction in soil (and land) quality can be assessed with an estimation of the recovery time needed to reverse these impacts, because in agriculture they are generally reversible.
- A new method to include the impacts on the life support functions of land in LCIA, based on Soil Organic Matter (SOM) has been presented. SOM has been detected as a good indicator for soil quality, and thus for life support functions.
  - The SOM indicator is consistent and sensitive enough only in the mid- and long-term impact assessment (crop rotation studies of above 10 years).
  - SOM is adequate for the comparison of the agricultural stage of different agricultural systems, while other life cycle stages will generally require other indicators. Indeed, land use impacts from agriculture are significantly different than those from other land uses. This fact, added to the global importance of agriculture for land degradation, requires for a specific approach to include land use impacts from agriculture in LCA.
  - Further aggregation of the different aspects of land use impacts is a matter of valuation, and has not been addressed in the present dissertation.

    - The methodology suggested has proved to be representative, consistent and applicable. Also, it correctly deals with the site-dependency required by agricultural LCA.
- To be applicable the SOM indicator requires the use of SOM models. The application of such models is data intensive, but the type of data required is generally available in agricultural LCA. Some practice is also required for the running of such models, although recent versions tend to be user-friendlier.
- The indicator soil organic matter does not correctly represent salinisation impacts, which should be addressed separately in the cases where this may be relevant. Soil loss through erosion is not represented either, although this may be included with proper methods dealing with the resource aspect of soil.
- The SOM indicator also has implications for the impact category Global Warming, which should be considered in a sound way by the LCA practitioner. Accordingly, SOM depletion by agricultural practices should be considered as carbon emissions in the agricultural stage, while SOM increases imply carbon sequestration that might be considered as a negative emission from agriculture.

Finally, the relative valuation of the different aspects affected by land use should be investigated, possibly on a case-by-case approach. Otherwise, a fair comparison between land-based products (e.g.: produced by agriculture or forestry) with petrol- or mineral-based ones will not be possible. Many applications of LCA lie within this scope (e.g.: biodiesel vs. diesel; cotton vs. polyester; wooden furniture vs. metal, etc.), and are accepting trade-offs that are not properly addressed. This remains a matter of further research, as it was not within the scope of this thesis.

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CHAPTER V. CONCLUSIONS AND OUTLOOK

"On pourrait donc dire que dans cette oeuvre j'ai seulement pris quelques fleurs cueillies par d'autres personnes, et ai composé un bouquet en y ajoutant le cordon"[1]
Montaigne

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In these conclusions, the issues dealt with in the different chapters are discussed (section V.1). Then, further research needs are suggested (V.2).

[1] One could thence say that in this work I have only taken a few flowers that had been gathered by others, and made a bunch of them by adding the string that joins them.

V.1. Conclusions

Twelve million hectares of arable land are lost every year in the world through erosion and land degradation, for which agricultural mismanagement is responsible to an important extent. The inclusion of such impacts in agricultural LCA is thus of crucial importance. The conclusions from the research presented in this thesis may be grouped under the categories of agricultural LCA practice, LCIA methodology, and practical application of LCA.

Agricultural LCA practice. Site-dependency in agricultural LCA

A relevant contribution from the thesis to the application of LCA to agricultural systems is the importance of site-dependency factors: site conditions -soil and weather- and farmers' practices.

1. **Farmer's practices are one of the main aspects of site-dependency**. It has been suggested for a long time that agricultural systems might be more site-dependent than industrial ones due to the physical site conditions (soil and weather). This research has confirmed that also different agricultural practices (farmers' technique) exert a considerable effect on the results. For instance, Chapter III has shown that the same field operation (e.g.: mowing, thinning, pruning, harvesting…) performed by different farmers results in variances of 30-50% in energy consumption. The influence of site-dependency to the credibility of the results of agricultural LCA has thus been evaluated. At the LCA practitioner's level, this means that inventory data collection has to take the local conditions (soil and weather) into account and incorporate the farmer's practices in a detailed way.
2. **LCA needs to be integrated with other tools of environmental analysis to allow for site-dependency**. LCA has been commonly understood as a methodology dealing with potential impacts, independent from the point of occurrence of the impacts. A clear separation from other disciplines has thus been drawn. However this thesis illustrates that the incorporation of methodologies commonly used in other disciplines (risk assessment, ecosystem modelling, etc.) into LCA increases the relevance of the results by incorporating a site-dependent analysis.

LCIA methodology for agricultural LCA. Impacts on life support functions of land

The most original contribution from this thesis lies in the context of the impact assessment methodology for land use impacts caused by agriculture.

3. **Agriculture exerts important occupation impacts on land**. Until now, land use impact assessment methods for LCA had been focussed on the assessment of transforming types of land use (i.e.: irreversible impacts); mainly indicators based on biodiversity have been proposed. In contrast, occupation impacts are possibly the main impacts from agricultural land use (640,000 km^2 of land degraded due to agricultural mismanagement in Europe), and affect biodiversity and the life support functions of land. This thesis gives evidence for the need of addressing occupation impacts and soil degradation in agricultural LCIA.

4. **Methods for the impact assessment of life support functions may be based on SOM**. Other authors have suggested indicators for life support functions (LSF) related to yield (fNPP, NPP) and soil parameters (organic matter, compaction, etc.), but no consistent method was available for their assessment. Chapter IV gives evidence of the representation of LSF by soil organic matter (SOM), and proposes an applicable, consistent and site-dependent method for the analysis. With this new methodology, agricultural systems may be compared, so sound decisions can be made on the environmental preference of different management techniques. SOM evolution has proven to be sensitive to the effects of different cropping practices and is thus able to predict mid- and long-term effects of farming types on soil quality.
5. **SOM models are a valid source of information in land use impact assessment for LCIA**. Previous authors suggesting SOM as a relevant indicator for life support functions pointed at approximations for data based on e.g.: measures of the amount of organic matter added to the field or direct measurements of SOM, which was rather impractical. Chapter IV demonstrates that mechanistic SOM models, available for a variety of regions of the globe, may be used to get the data needed for the assessment of impacts on life support functions. These models are data-intensive but may be applied by the LCA practitioner with some help from agronomists.
6. **SOM evolution trend may guide agricultural practices**. As most process-based models, the accuracy in the predictions from SOM models is not high, but the mid-term trend in SOM evolution may aid decision-making in agricultural LCA. Chapter IV shows how this trend is obtained by running the models for 20-30 years. Agricultural management practices may thus be characterised according to their likely effects on the soil quality and guide farmers consequently.

Applications of agricultural LCA. Eco-labelling and benchmarking

Typical applications of agricultural LCA are the description of systems in order to detect environmental hotspots (e.g.: for the establishment of eco-labelling criteria), or the comparison of products and technologies, either for benchmarking, definition of good agricultural practices (GAP), etc. If organic agriculture is not only aimed at preserving human health and ecosystem's quality at the local level of the site, but also the environmental quality on a broader sense, the contribution of LCA to the definition of best practices will be crucial. Besides, new environmental issues will have to be considered when assessing agricultural technologies with a holistic perspective.

7. **Energy consumption greatly determines agriculture's environmental sustainability**. Criteria for organic agriculture have usually focused on the nature of the substances introduced in the field, such as fertilisers and pesticides, possibly due to the ease of obtaining data for these criteria. Nevertheless, the results obtained in this thesis highlight the importance of energy consumption by agriculture, which may give higher impacts for organic farming when compared to integrated farming as it has been shown in Chapter III. Energy consumption is found to be significantly higher in organic farming than in integrated farming in apple production in New Zealand, and it contributes above 50% to most impact

categories considered in the study. Therefore, holistic approaches such as LCA covering the different environmental impacts affected by agriculture should be promoted when designing certification schemes or assessing the environmental soundness of agriculture.

8. **Criteria for agricultural certification schemes should incorporate site-dependency**. The general objective of agricultural certification schemes (e.g. organic or integrated agriculture) is to prevent the loss of nutrients and agro-chemicals to affect life, both human and of the rest of species. Besides, a reference has been found in the literature suggesting that LCA applications to eco-labelling should be site-independent. Nevertheless, this thesis has shown that the same input (resource consumption) produces very different outputs (emissions) in different sites. Indeed, Chapter III presents a case where the use of a particular pesticide results in small emissions in some sites, and is thus accepted by the integrated farming scheme; the same ingredient applied in a sensitive area leads to emissions up to one order of magnitude higher due to soil and weather conditions. Therefore indicators based on actual emissions (both to air and groundwater) should incorporate site conditions and farmer's practices and then be used in certification schemes. Bearing in mind the diverse reality of agricultural production in the world, a broader and site-dependent perspective should be considered when revising the criteria for organic agriculture.
9. **Comparative agricultural LCA should incorporate occupation land use impacts**. Benchmarking of good agricultural practices (GAP) should include land occupation impacts. Besides, GAP should be defined on a site-dependent basis, because this thesis demonstrates that a sound practice in one place may not be so in another due to soil and weather conditions.

V.2. Outlook: a young field of research

Agricultural LCA is a young discipline, particularly in Southern Europe and outside Europe. I hope this thesis has served to detect and outline possible new lines of research. Agronomic research institutes should now take over and include agricultural LCA in their agendas. Several international projects are currently being developed on the subject, but many methodological and practical aspects still need further refinement. Particularly, the following issues deserve special attention when designing research programmes:

- **Holistic environmental assessment of farming systems**. The sustainability of different types of agriculture (e.g.: organic; integrated; conventional) must be explored from a holistic perspective, in order to incorporate other environmental aspects into the debate: impacts from energy consumption, fertiliser use and impacts on land use (in addition to land use efficiency). Agricultural LCA may be used and refined to cope with this analysis. In the European context, this should be included in the mid-term review of the Common Agricultural Policy, which according to the EU Strategy for Sustainable Development[2] "should reward quality rather than quantity by, for example, encouraging the organic sector and other environmentally-friendly farming methods".
- **Revision of agricultural certification schemes**. Organic farming is usually supported by claiming that it brings about positive effects on the environment and on rural development. However it is also suggested that other farming systems might have fewer detrimental effects, as it might be suggested by the results from Chapter III. A transparent assessment of the trade-offs between different farming systems is thus needed, and can be provided by agricultural LCA. A direct application of such studies would be a revision of the scope of agricultural certification schemes.
- **Machinery data needs in agricultural LCA**. Databases for agricultural machinery adapted to local conditions need to be developed for an easier and more consistent application of agricultural LCA. Particularly the effect of total use in lifetime and the effects of farmer's practices on machinery durability should be explored and clarified in these databases.
- **Need for development of soil organic matter databases**. Extensive databases of SOM evolution under different cropping practices should be developed in order to facilitate the application of this indicator through simplified characterisation factors. Public bodies may facilitate this by including organic matter as an indicator of land use sustainability in existing monitoring programmes thus providing databases with SOM levels in different regions.
- **Assessment of the relationship between soil quality and other impact categories**. Impact categories have generally been regarded as quite independent ones from the others, but the work with SOM models has shown how the development of an indicator for soil quality can also be connected to other environmental aspects. Accordingly, the practical effects of using SOM models for other impact categories (mainly global warming

[2] COM/2001/0264: *A Sustainable Europe for a Better World: A European Union Strategy for Sustainable Development* (Commission's proposal to the Gothenburg European Council).

and nutrification) should appear in LCA research. For instance, this might clarify the treatment of carbon fixation by agricultural systems in relation to global warming.

- **Trans-disciplinary approach for other environmental issues**. The inclusion of locally relevant environmental problems such as salinisation and erosion into LCIA should be developed. This will possibly require a closer integration of LCA with other disciplines and tools for environmental analysis, in the line shown in this thesis.
- **Valuation of different aspects of land use impacts**. A further application of agricultural LCA is the comparison of "petrol- or mineral-based" products with "land-based" ones. It is obvious that the types of impacts affected by these products lie within different areas of protection, and trade-offs exist in any election between them. Accordingly, valuation methods should be developed for such comparisons.
- **Farmer's training and awareness**. The communication of LCA results to farmers must be done in a way that seeks their complicity and acknowledges the need of covering their socio-economical aims as well as the improvement of their environmental performance. Economic incentives may be studied with this purpose, such as the cross-compliance measures designed in the scope of the EU's Common Agricultural Policy Mid-Term Review[3].
- **Raising consumer's awareness on global impacts from agriculture**. Consumers' awareness on the environmental consequences of agriculture should be broadened beyond the toxicity issues. In relation to this, the European Commission has recently started a consultation with European consumers in order to draw an action plan for the promotion of organic agriculture. Possibly, the use of Environmental Product Declarations based on LCA results may help in the communication of multidimensional environmental impacts.

The sustainability issue is very complex because of its multidimensional nature. In this thesis, I've incorporated methodologies from other disciplines into LCA, in order to embrace the multi-faceted aspects of agricultural impacts on the environment. I hope that this work and its results will encourage research centres focussing on agriculture's environmental performance to use such holistic approaches, contributing in this way to understand the role of our stewardship of agro-ecosystems.

[3] COM(2002)394 final. *Communication from the Commission to the Council and the European Parliament: Mid-Term Review of the Common Agricultural Policy*. Brussels, 10.7.2002.

In the development of this thesis I've used around 4.5 kg (quick and dirty estimation) of paper for drafts, discussions, etc. Most of it was re-used waste paper from spare photocopies, page proofs, etc.; in some occasions also recycled paper was used, and for the final drafts white paper was used 2-sided. In addition, a laptop (my good and old Borja Mari) was used at least 30% of its time to the development of the thesis for 3.5 years. The thesis has been developed throughout the world, and plane flights to England (Barcelona–London–Barcelona), Sweden (Barcelona–Brussels–Gothenburg–Brussels–Barcelona), Denmark (Barcelona–Zürich–Copenhagen–Stuttgart–Barcelona) and New Zealand (Barcelona–Rome–Seoul–Auckland–Christchurch– and back! with an additional trip from Christchurch to Sydney) should be added to the list. Also the travels for presentations in congresses may be considered: Barcelona–Bordeaux by car; Barcelona–Berlin by plane, and then train to Leipzig; Barcelona–London again, and then by train to Brighton; Barcelona–Brussels two or three times; Barcelona–Brussels–Gothenburg; Barcelona–Madrid by train… Obviously, feeding myself for all this time has not been an energy-saving task! In summary, all these impacts add to the environmental burdens of my thesis, which, approximately and without the feeling of being inaccurate, account for A LOT. I sincerely hope that the suggestions for environmental improvement I've done with the thesis will some day be used to help reducing the environmental burdens of agriculture, in order to counterbalance those of the thesis! Luckily enough, the development of the thesis has provided much more functions than the mere dissertation…

The thesis has been written while listening to (mainly) Jean-Michel Jarre, Paco de Lucía, Mike Oldfield, Nick Drake, Moby, the ultra-lounge collection, Ismael Lö and many others. Thanks to all for your inspiration!