Integrating Biodiversity and Ecosystem Services in Life Cycle Assessment: Methodological Proposals for New Challenges

Ricardo F.M. Teixeira

Research Group of Plant and Vegetation Ecology, Department of Biology, University of Antwerp, Campus Drie Eiken, Universiteitsplein 1, B-2610 Wilrijk, Belgium
ricardo.teixeira@uantwerpen.be

Life Cycle Assessment (LCA) is a method used to quantify potential environmental impacts in the entire life cycle of a product or service, starting at raw material acquisition, production, use, and eventually its disposal. It is the leading methodology for environmental metrics today – the French government recently ran a pilot for a labelling program for consumer goods where indicators displayed in labels should be calculated using LCA. The European Commission did the same for their Product Footprint Guidelines.

However, LCA features many methodological shortcomings when dealing with biophysical indicators. Land use, biodiversity and ecosystem services are relevant aspects to policy-makers and consumers, but their characterization factors are not yet accurately calculated. LCA has a hard time grasping effects that are dynamic, scale-dependent, non-linear and/or hard to quantify unambiguously. In fact, out of the four conventional types of ecosystem services (provisioning services, regulating services, cultural services and supporting services) only the first and the last are partially captured in LCA results. Similarly, out of the five drivers of change to biodiversity identified by the Millennium Ecosystem Assessment (habitat change, climate change, pollution, invasive alien species and overexploitation) only the first three are included in standard LCA calculations.

This review article presents a brief assessment of how biodiversity and ecosystem services are incorporated in LCA today. The standard Life Cycle Impact Assessment models are described according to the most recent developments. Ground-breaking work is undergoing, making the field very active and constantly innovating. For example, regionalized characterization factors for the impacts of biodiversity are now available using species richness as a proxy for the overall state of biodiversity. Tools for ecosystem services have also been proposed. Finally, a compilation of some challenges ahead is listed to present several ideas for the improvement of current impact assessment methods.

1. Introduction

Life Cycle Assessment (LCA) started out in the late 1960's as a metrics tool for energy efficiency of facilities and products (Guinée et al., 2011). It matured in the ensuing decades to encompass resource use, pollution control, and solid waste. In the 1990's LCA was eventually standardized in the ISO 14040 series. Today LCA can be defined as a tool aiming to evaluate potential impacts of the production, use, and waste management of goods (Tukker, 2000). The European Commission (EC) introduced life cycle thinking through the Integrated Product Policy in COM/2003/0302. By that time, LCA was already widely adopted to assess environmental damages, initially for research purposes. During the early 2000's LCA started supporting management decisions in companies. Today, LCA is used in benchmarks or comparisons in the framework of investment decision, product development or contracting subcontractors. The French government recently ran a pilot for a labelling program for consumer goods where indicators displayed in labels should be calculated using LCA (ADEME-AFNOR, 2013). The European Commission (EC) did the same for their Product Footprint Guidelines (EC, 2013) to calculate environmental indicators for assertive comparisons between products and organizations. Both labelling schemes may be a reality in

a near future, establishing LCA in private companies as an environmental management tool. After four decades of methodological development, LCA is a crucial part of the transition to a green economy, and the privileged method for calculations of environmental impacts.

Due to its historical roots, LCA is particularly suited to account for inflows and outflows of substances from processes and products, i.e., for substance emissions. That brings limitations in scope and usability (Freidberg, 2013). LCA does not deal well with effects that are (1) dynamic, i.e. non-constant in time, (2) scale-dependent, i.e. vary in location, (3) non-linear, e.g. mutually reinforcing impacts in different categories, and (4) hard to quantify unambiguously (Curran et al., 2011). Besides, the results of one LCA are closely connected to the functional unit, which is an unequivocal description of the use pattern of the product or service. For non-linear impact categories, the marginal contribution of one more product (or one more unit of use) may be zero, but the impact profiles in LCA are continuous, which means that some impact will always be interpolated. Land use, water consumption, ecosystem services and biodiversity are examples of impact categories that combine all these features. Impacts to biodiversity pose a particularly difficult challenge to LCA (Freidberg, 2013). Biodiversity is site-specific (Geyer et al., 2010), and LCA's simplified, high-level approach has problems modelling it across product life cycles (Curran et al., 2011). Ecosystem services are similar as phenomena – complex, dynamic and non-linear effects of multiple causes. Virtually all life cycles, including products in the chemical sector, involve at some tier land operations and emissions leading to changes in the state of biodiversity; however, it is particularly important to take biodiversity impacts into consideration in LCA studies dealing directly with land use and land use change (such as food or bioenergy studies, as in Čuček et al., 2011) – which currently is scarcely done due to methods not being recognized as stable and mature (Freidberg, 2013).

The present is a review article aiming at presenting a very short overview of how biodiversity and ecosystem services are currently included in LCA. Next we discuss the concepts of "biodiversity" and "ecosystem services".

2. Biodiversity and ecosystem services

2.1 Definitions

Ecosystem services are defined as benefits to human societies provided by ecosystems (Pereira and Cooper, 2006). They can be divided into four groups: (1) provisioning services (products obtained from ecosystems such as food or fresh water); (2) regulating services (benefits obtained from regulation of ecosystem processes such as water purification or pollination); (3) cultural services (non-material benefits such as recreation and eco-tourism); and (4) supporting services (those necessary for the production of all other ecosystem services, such as soil formation or nutrient cycling).

Biodiversity can be defined as "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (MA, 2005). There are four levels of hierarchical components in biodiversity: (1) genes, (2) species, (3) communities, and (4) ecosystems and landscapes. Each level has specific compositional, structural, and functional attributes (Noss, 1990).

Biodiversity has a double role as promoter of and benefitted by ecosystem services. For example, local population abundance of species can ensure the provision forest foods, pest control, or bird watching (Pereira and Cooper, 2006); it can also be promoted by other ecosystem services such as provision of habitat.

2.2 The Millennium Ecosystem Assessment

The Millennium Ecosystem Assessment (MA) was an initiative by the United Nations to assess how human well-being is influenced by changes in ecosystem services. The MA determined the past trends, present (2001-2005) and future scenarios for ecosystem changes and ensuing impacts at a global and several regional scales. Since the program was designed to respond to data needs of four conventions, namely the Convention on Biological Diversity, the United Nations Convention to Combat Desertification, the Ramsar Convention on Wetlands, and the Convention on Migratory Species, biodiversity was a key aspect talked by the MA. The MA ascribes an intrinsic value to biodiversity and considers it to have untapped option values for the future. Moreover, biodiversity is a capital asset in the sense that it provides provisioning, regulating, and ecosystem services to society and contributes to many economic activities.

A body of science was thus compiled that justifies and illustrates how biodiversity is directly affected by human activities, very often negatively (MA, 2005). The five major drivers of biodiversity change in ecosystem services were (Figure 1): (1) habitat change, (2) climate change, (3) invasive alien species, (4) overexploitation of species, and (5) pollution (MA, 2005). The outlook is clearly negative, and few trends representing decreasing impacts can be found. This worrisome fact caused the recent Rio+20 Summit to

shift the focus of the sustainability agenda to biodiversity, as it is the cornerstone of sustainable development (Earth Summit, 2012).

The MA qualitatively determined the key elements inflicting damage to biodiversity as a first step to finding metrics to determine the magnitude and scale of those impacts. While ecosystem services such as food are monitored, most of the others are not, including those related to biodiversity (Pereira and Cooper, 2006). Such task is imperious but difficult. The MA itself recognized that biodiversity is a multidimensional problem that "poses formidable challenges to its measurement", and that the most common measuring techniques (direct measuring of species richness or genetic diversity) do not capture the whole picture (MA, 2005). Issues such as "variability, function, quantity, and distribution" are usually disregarded.

3. Biodiversity and ecosystem services in LCA

3.1 Simplified metrics using expert judgement

The most simple and direct way to consider biodiversity uses expert judgement to grasp the complexity of biodiversity dynamics. Since the effects on biodiversity respond to multiple drivers and are hard to determine analytically, but there is a myriad of studies dealing with particular aspects and key causal relations, this approach can congregate the experience of experts and qualitatively determine predictable changes to the state of ecosystems. Under this direct approach qualitative estimations are usually converted as a second step to quantitative indicator of species number or diversity, eventually as a gain or loss function. The most well-known method is the Swiss Agricultural LCA, or "SALCA-Biodiversity" (Jeanneret et al., 2008). Some authors also propose the use of expert judgment to answer questions and attribute subjective rankings, in terms of the impact to biodiversity, to unit processes in LCA (e.g. Penman et al., 2010). For example, an impact of -10 would be attributed to the process "forest clearing", and an impact of +10 to the process "afforestation".

The main limitation in this procedure is that it is not systematic. Biologists typically choose quantitative indicators to assess the state of biodiversity, such as number or diversity of species, usually compiled in indices (Duelli and Obrist, 2003). Those indicators can then be used to assess the general state of ecosystems (Moonen and Bàrberi, 2008). These biodiversity indicators are subject to experimental conditions and are not easily extrapolated in an LCA setting, which uses a top-down approach to modelling. In LCA, they would be a function of unit processes and not unit flows. The indicators associated with unit processes are usually a function of land uses or farming practices, not so much of substance emissions or resource use, which are the major part of life cycle inventory data.

3.2 Quantitative endpoint methods

Quantitative methods were developed to address the need to tailor biodiversity indicators to LCA inventories and also for consistency purposes regarding other life cycle impact assessment methodologies for other impact categories. Thus far biodiversity is mostly included in LCA as an endpoint indicator (calculated from midpoint impact categories), as concluded by Curran et al. (2011) in a recent review. Biodiversity is calculated as a result of the impact categories of land use, water use, climate change, acidification and eutrophication, and ecotoxicity. Together, these five categories concur to 3 of the 5 main drivers of biodiversity loss identified by the MA. Invasive species and overexploitation are not included in any current method to assess biodiversity in LCA. Due to its importance, as identified by the MA, and since it is the most active field today, the rest of this article is devoted to biodiversity impacts from land use only.

Unlike quantitative methods, which consider "biodioversity" as an aggregated indicator that combines the different characteristics of the phenomenon, quantitative methods require an operational definition of biodiversity that can be quantified. That definition is usually to consider number of species, or species richness, as a representation of the state of biodiversity as a whole.

The most frequently used indicator for biodiversity loss in LCA (in particular, due to land use) is thus the "potentially disappeared fraction (PDF) of species" (unit less). This measurement expresses an effective loss of habitat: PDF = 1 would mean a total loss of habitat value for biodiversity. Biodiversity endpoints thus obtained are measured in $PDF.m^2.y$ – the fraction of species potentially lost in an area and time frame. Originally, only vascular plants were used in calculations. Vascular plants are used in biometrics since they are the main primary producers in terrestrial ecosystems, and because the diversity of plants is one of the best available predictors of diversity of other taxa (Pereira and Cooper, 2006).

Two crucial aspects of this model require particular care: the reference or baseline against which the PDF of species is determines, and how to estimate species dynamics as a function of area and time.

3.3 Simplified description of the impact assessment model

The variation of species richness with time is captured using the framework originally proposed for changes in land quality by Lindeijer et al. (2002) and depicted in Figure 1. The model was later generalized

by Mila-i-Canals et al. (2007) and was finally adopted to account for biodiversity impacts from land use by de Schryver et al. (2010), who used it to calculate characterization factors. ReCiPe, the most widely used LCA methodology today, makes use of these characterization factors. The model in Figure 1 assumes that land occupation is a one-time, on/off phenomenon during a given time (t) and in a given area (A) that originates an instantaneous decrease in quality (Q) – or increase in loss of quality (-Q) – in area A at t_0. This decrease is sustained until the new occupation is abandoned at t_1. The ecosystem then enters a relaxation period during which it regains quality. Q may return to the original level $-Q_0$ (at t_2) or stabilize at a lower Q-level of $-Q_f$.
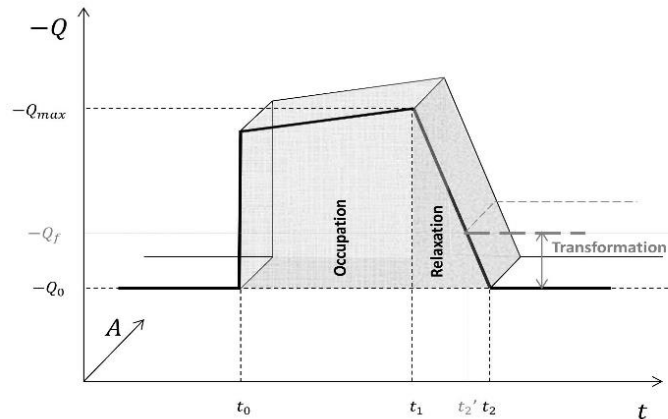

Figure 1: Impacts from land use change on ecosystem quality (adapted from Mila-i-Canals et al., 2007). An environmental pressure generates temporary decreases in quality during the time of occupation and relaxation. When the final and initial ($-Q_f$ and $-Q_0$) levels are different there is also a permanent loss of quality. Variables: -Q – decrease in quality; A – area, t – time; Subscripts: 0 – start of occupation; 1 – end of occupation; 2 – end of relaxation; f – final level; max – maximum level.

Figure 1 shows how ecosystem quality varies with time, but the variable Q needs to be operationalized using a different model – in this case a model that defines Q, relating it to biodiversity, and also what is the functional form that relates Q and A. There is one study that focuses on functional diversity (Souza et al., 2013), but Q is usually understood in this context as a synonym for species richness. The simplest macroecological model that relates species richness and area is known as the Species-Area Relationship (SAR) (Sarkar and Margules, 2002), also known as the Arrhenius (1921) model, shown in Eq(1).

$$S(A) = (A/A_o)^z \times S_o \quad (1)$$

$S(A)$ is the number of species in area A and S_o the number found in the entire biome of area A_o before the occupation change. The parameter z is a constant in the range of 0.10-0.35, depending on the taxon and scale. This article does not present a detailed analysis of this model, its limitations and suggestions for future developments – which can be found in Teixeira (2014).

4. Future methodological developments

In recent years there has been an intensification of ground-breaking work towards including biodiversity and ecosystem services in LCA. For example, regionalized characterization factors for the impacts of land use on biodiversity are now available using species richness as a proxy for the overall state of biodiversity (Baan et al., 2013). An alternative approach using ecosystem scarcity has also been applied successfully to New Zealand (Coelho and Michelsen, 2014). Comprehensive tools for ecosystem services have also been proposed (Zhang et al., 2010) together with assessments for particular ecosystem services. The methodology to determine the variation in ecosystem services is similar to that of biodiversity impacts – it is the indicator used to quantify Q that varies. According to Koellner et al. (2013), "consolidated" methodologies exist for five ecosystem services: (1) biotic production potential, using as indicator the deficit of Soil Organic Matter due to land use (Brandão and Milà i Canals, 2013); (2) climate regulation potential, using carbon flows change due to land use as an indicator (Müller-Wenk and Brandão, 2010); (3) freshwater regulation potential, measured as ground water recharge rate; (4) erosion regulation potential, using erosion resistance; and (5) water purification potential, using cation exchange capacity as an indicator (services 3-5 in Saad et al., 2013).

The acceptance of these methods by the LCA community has been meagre. The EC did not include biodiversity as an impact category in the Product Footprint Guidelines (EC, 2013) for two reasons: they avoided endpoint categories, and they considered that the method was not sufficiently developed. The French labelling program (ADEME-AFNOR, 2013), on the contrary, wanted to make biodiversity a mandatory indicator for some products (such as food), but there is still confusion on how to measure it (Freidberg, 2013). Some Non-Governmental Organizations (NGOs) are also highly suspicious of the capacity of LCA to capture impacts that are inherently site-specific and complex (Freidberg, 2013).

It is clear that there is a long road ahead before any method can be considered "mature". Experts from other areas – Biology, Ecology, and also NGOs, who understand what happens on the field – must be included in the discussion to help with methodological responses to new challenges raised by recent developments.

Some examples of open questions that potentially compromise the accuracy of the methodology proposed are the following. There is a conceptual issue with biodiversity that concerns its dual nature as having existence value (species richness as an end in itself) and use value (diversity as a natural resource). So far models consider mostly the existence value of biodiversity, but in some low-diversity ecosystems some species may be key to sustain ecosystem services and thus have a high use value regardless of the total number existing in that ecosystem. Landscape effects are neglected when using species-level indicators, but fragmentation and connectivity, for example, are crucial to understand diversity dynamics (Curran et al., 2011). Current methods also need to increase bio-geographical differentiation (taxonomic and geographic coverage), and carefully equate the scale of the models (Koellner et al., 2013). So far, the eco-region approach has been followed, but the exact same models used at a country scale could yield different results because ecological models like the SAR are scale-dependent (Teixeira, 2014).

The model described by Figure 1 is a simplification that implies that there is an immediate loss in biodiversity after occupation. It also implies that the impacts of occupation and regeneration depend only of the potential Q level and the type of occupation. The land use history is a crucial variable in all these effects (Koellner et al., 2013). Further, it is unclear how engineered systems are depicted by such a framework. Active restoration using artificial land systems can have different results from natural re-vegetation (for one example of an engineered system of pastures that can improve soils and support biodiversity, see Teixeira et al., 2011). The comparison of all current land uses with an idealized natural Q level is also of debatable usability, and an alternative approach could be to use some land use configuration in a past year (e.g. 2000) as a baseline (Koellner et al., 2013).

5. Conclusion

Due to LCA's importance today, finding a way to incorporate biodiversity and ecosystem services in its standard framework is of the upmost importance. This is particularly relevant for sectors such as agri-food and forestry or bioenergy, but it has implications for all products – including chemicals. Interesting developments were recently proposed, but there is still a significant number of challenges to address that raise the LCA community's suspicion about the maturity of current methods. This area of research rests on the frontier between Biology, Ecology and Industrial Ecology – but most developments have traditionally come from within the LCA community. It is crucial to broaden the range of experts developing methods, particularly involving biologists and ecologists in the discussion. For now, it is yet to be determined whether any quantitative measure will accurately depict the state of biodiversity as it is measured "on the ground" (Freidberg, 2013, quoting a Rainforest Alliance representative).

Acknowledgements

References

Arrhenius, O., 1921, Species and area, J. Ecol. 9, 95-99.

ADEME-AFNOR (French Environment and Energy Management Agency - Association Française de Normalisation), 2013, Environmental labeling of consumer products (in French), <affichage-environnemental.afnor.org> accessed 01.06.2013 (in French).

Baan, L., Alkemade, R., Koellner, T., 2013, Land use impacts on biodiversity in LCA: a global approach, Int. J. LCA 6, 1216-1230.

Brandão, M., Milà i Canals, L., 2013, Global characterisation factors to assess land use impacts on biotic production, Int. J. LCA 18, 1243-1252.

132

Coelho, C.R.V., Michelsen, O., 2014, Land use impacts on biodiversity from kiwifruit production in New Zealand assessed with global and national datasets, Int. J. LCA 19, 285-296.

Čuček, L., Klemeš, J.J., Varbanov, P., Kravanja, Z., 2011, Life Cycle Assessment and Multicriteria Optimization of Regional Biomass and Bioenergy Supply Chains. Chemical Engineering Transactions, 25, 575-580 DOI:10.3303/CET1125096.

Curran, M., De Baan, L., De Schryver, A.M., Van Zelm, R., Hellweg, S., Koellner, T., Sonnemann, G., Huijbregts, M. J., 2011, Toward meaningful end points of biodiversity in life cycle assessment, Environ. Sci. Technol. 45, 70-9.

De Schryver, A. M., Goedkoop, M. J., Leuven R. S. E. W., Huijbregts, M. J., 2010, Uncertainties in the application of the species area relationship for characterization factors of land occupation in life cycle assessment, Int. J. LCA 15, 682-691.

Duelli, P., Obrist, M. K., 2003, Biodiversity indicators: the choice of values and measures, Agr. Ecosyst. Environ. 98, 87-98.

Earth Summit, 2012, The Future We Want. Final declaration of the Rio+20 Summit, Rio de Janeiro, Brazil.

EC (European Commission), 2013, Product Environmental Footprint (PEF) News <ec.europa.eu/environment/eussd/smgp/product_footprint.htm> accessed 13.12.2013.

Freidberg, S., 2013, Calculating sustainability in supply chain capitalism, Econ. Soc. 42, 571-596.

Geyer, R., Stoms, D.M., Lindner, J.P., Davis, F.W., Wittstock, B., 2010, Coupling GIS and LCA for biodiversity assessments of land use, Int. J. LCA 15, 454-467.

Guinee, J.B., Heijungs, R., Huppes, G., 2011, Life cycle assessment: Past, present, and future, Environ. Sci. Technol. 45, 90-96.

Jeanneret, P., Baumgartner, D. U., Knuchel, R. F., Gaillard, G., 2008, Integration of biodiversity as impact category for LCA in agriculture (SALCA-Biodiversity). Proceedings of the 6th International Conference on LCA in the Agri-Food Sector, November 12-14, Zurich, Switzerland

Koellner, T., Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., Milà i Canals, L., Saad, R., Souza, D. M., Müller-Wenk, R., 2013, UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA, Int. J. LCA 18, 1188-1202.

Lindeijer, E., 2000, Biodiversity and life support impacts of land use in LCA, J. Clean. Prod. 8, 313-319.

MA (Millennium Ecosystem Assessment), 2005, Ecosystems and Human Well-being: Biodiversity Synthesis of the MA. World Resources Institute, Washington, DC, USA.

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Knuchel, R.F., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., 2007, Key Elements in a Framework for Land Use Impact Assessment Within LCA, Int. J. LCA 12, 5-15.

Moonen, A.C., Bàrberi, P., 2006, Functional biodiversity: An agroecosystem approach, Agr. Ecosyst. Environ. 127, 7-21.

Müller-Wenk R., Brandão M., 2010, Climatic impact of land use in LCA – carbon transfers between vegetation soil and air, Int. J. LCA 15, 172-182.

Noss, R. F., 1990, Indicators for Monitoring Biodiversity: A Hierarchical Approach, Conserv. Biol. 4, 355–364.

Penman, T. D., Law, B. S., Ximenes, F., 2010, A proposal for accounting for biodiversity in life cycle assessment, Biodivers. Conserv. 19, 3245–3254.

Pereira, H., Cooper, D., 2006, Towards the global monitoring of biodiversity change, Trends Ecol. Evol. 21, 123-129.

Saad R., Koellner T., Margni M., 2013, Land use impacts on freshwater regulation, erosion regulation and water purification: a spatial approach for a global scale, Int. J. LCA 18, 1253-1264.

Sarkar, S., Margules, C., 2002, Operationalizing biodiversity for conservation planning, J. Biosci. 27, 299-308.

Souza, D. M., Flynn, D., Rosenbaum, R.K., Lisboa, H. M., Koellner, T., 2013, Land use impacts on biodiversity: proposal of characterization factors based on functional diversity, Int. J. LCA 18, 1231-1242.

Teixeira, R., 2014, Which species count? Reflections on the concept of species richness for biodiversity endpoints in LCA, Chemical Engineering Transactions, 42, 133-138 DOI:10.3303/CET1442023.

Tukker, A., 2000, Life cycle assessment as a tool in environmental impact assessment, Environ. Impact Asses. 20, 435-456.

Zhang, Y., Baral, A., Bakshi, B. R., 2010, Accounting for ecosystem services in Life Cycle Assessment, Part II: toward an ecologically based LCA. Environ. Sci. Technol. 44, 2624-31.