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Scherer, L.A.; De Laurentiis, V.; Marques, A.; Michelsen, O.; Migoni Alejandre, E.; Pfister, S.; ... ; Rugani, B.

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COMMENTARY AND DISCUSSION

Linking land use inventories to biodiversity impact assessment methods

Laura Scherer[1] · Valeria De Laurentiis[2] · Alexandra Marques[3] · Ottar Michelsen[4] · Elizabeth Migoni Alejandre[1] · Stephan Pfister[5] · Francesca Rosa[5] · Benedetto Rugani[6]

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Abstract

There is generally a mismatch in the land use classification of life cycle inventory (LCI) databases and life cycle impact assessment (LCIA) methods. This mismatch can hinder the proper assessment of land use impacts on biodiversity. To facilitate such assessments, we matched the land use classes of two global LCIA methods to five widely used LCI databases, one LCI nomenclature, and one multi-regional input–output database. In unclear cases, we assumed the worst case. Assumptions were especially necessary for unspecified land use intensity classes. We conclude with recommendations for LCI database and LCIA method developers.

1 Introduction

In order to assess the impacts of land use on biodiversity within the life cycle assessment (LCA) framework, the land use inventory is multiplied with impact characterization factors. The elementary flows of the inventory and the characterization factors are specific to different land use types and, in some cases, land use intensities. However, there is often a mismatch in the land use classification between life cycle inventory (LCI) databases and life cycle impact assessment (LCIA) methods, which inconveniences LCA practitioners. First, it can increase the data preparation requirements of the assessment to achieve proper matching. Second, it may impair the robustness of the assessment, as different LCA practitioners may make different choices when matching the inventory with the LCIA methods and consequently get deviating results.

To overcome this hindrance, Koellner et al. (2013a) proposed a standardized land use classification. It consists of multiple levels of detail to provide flexibility to LCI database and LCIA method developers. Despite the good intentions, the problem remains. There are only a few examples where LCIA method developers aligned their characterization factors with the standardized land use classes or the land use classes of a selected LCI database (e.g., Alejandre et al. under review, Bos et al. 2020). The inconsistency among LCI databases and among LCIA methods compounds the problem. Here, we matched the land use classes of two global LCIA methods to several widely used LCI databases as part of the ecosystem quality task force of the Life Cycle Initiative hosted by UN Environment. This exercise aimed to make it easier for LCA practitioners to assess the impacts of land use on biodiversity and identify some issues for LCI database and LCIA method developers to consider for further improvements. Although we focus on biodiversity impacts, the matching can also be relevant to other impacts driven by land use, such as impacts on ecosystem services. A similar matching exercise, presented in Sanyé-Mengual

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Laura Scherer
l.a.scherer@cml.leidenuniv.nl

1 Institute of Environmental Sciences (CML), Leiden University, Leiden, The Netherlands

2 European Commission-Joint Research Centre, Via Enrico Fermi 2749, 21027 Ispra, VA, Italy

3 PBL Netherlands Environmental Assessment Agency, The Hague, The Netherlands

4 Department of Industrial Economics and Technology Management, Norwegian University of Science and Technology (NTNU), Trondheim, Norway

5 Institute of Environmental Engineering, ETH Zurich, 8093 Zurich, Switzerland

6 RDI Unit on Environmental Sustainability Assessment and Circularity (SUSTAIN), Environmental Research & Innovation (ERIN) Department, Luxembourg Institute of Science and Technology (LIST), 41 Rue du Brill, L-4422 Belvaux, Luxembourg

et al. (under review), considered more LCIA methods but fewer LCI nomenclature systems. The present paper thus complements existing efforts.

2 Methods

We performed the matching of land use classes between the nomenclatures adopted in land use inventories and LCIA methods manually. We considered the following five LCI databases, one LCI nomenclature, and one global multi-regional input–output (MRIO) database:

- Agri-footprint® (version 5.0), retrieved from https://www.agri-footprint.com/ and via private communication with Block Consultants
- Agribalyse (version 3.0.1), retrieved from https://doc.agribalyse.fr/documentation-en/
- ecoinvent[1] (version 3.4), retrieved from https://www.ecoinvent.org/
- EF nomenclature, the nomenclature system adopted by the Environmental Footprint (EF) reference package (version 3.0), retrieved from https://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml
- EXIOBASE (version 3.4), retrieved from https://www.exiobase.eu/index.php/data-download/exiobase3mon and Stadler et al. (2018)
- GaBi (2021 edition), retrieved via private communication with the Fraunhofer Institute for Building Physics IBP
- US LCI database (version FY21.Q1.01), retrieved from https://www.lcacommons.gov/nrel/search

As a comparison, Sanyé-Mengual et al. (under review) fully considered ecoinvent 3.6 and the EF3.0 nomenclature, while partially considering Agri-footprint.

The elementary flows of LCI databases and the EF nomenclature formed the basis for the matching. If annual and permanent crops were not distinguished, we would have examined the unit processes. This was not necessary for any of the databases considered.

The land use classes we considered for LCIA are a mix of those included in the characterization models of Chaudhary and colleagues (2016, 2018). Sanyé-Mengual et al. (under review) also included the model by Chaudhary and Brooks (2018). We selected the characterization model developed by Chaudhary et al. (2016), as it received an interim recommendation in an earlier phase of the Life Cycle Initiative (Mila i Canals et al. 2016; Jolliet et al. 2018). Further, we selected the characterization model by Chaudhary and Brooks (2018), which built on the earlier model, as they additionally considered land use intensities, which was one of the areas for improvement pointed out by the Life Cycle Initiative (Jolliet et al. 2018). The two models also differ in that Chaudhary and Brooks (2018) subdivided forestry into managed and plantation forests but merged annual and permanent crops to cropland. In our mix of land use classes, we mostly followed the classes by Chaudhary and Brooks (2018) but distinguished annual and permanent crops as done by Chaudhary et al. (2016). Since agriculture is the biggest anthropogenic land use driver, it seems valuable to provide more specificity than a generic cropland class. The mix also covers the land use classes considered in the method by Kuipers et al. (2021) that assesses the impacts of land fragmentation.

As a guide for the matching of land use classes, we used Table S1 "Land classification" and Table S7 "Matching LU flows" from Chaudhary and Brooks (2018) and Appendix A3 "Land use classification for LCA applications" from Chaudhary et al. (2015). The latter developed the predecessor method of Chaudhary et al. (2016) that is limited to vertebrates and does not yet cover plants for global species loss. Where the land use classes of the land use inventories could not clearly be linked to any of the descriptions in those supplementary tables and we had to make assumptions, we indicated this uncertainty with gray font color. Since a different person matched each database or nomenclature, we discussed such potential subjectivity-based uncertainty cases during group consultations to remain consistent throughout the matching.

For unspecified or aggregated elementary flows in any databases or nomenclatures, we assumed the worst-case scenario, similar to the approach taken by Sanyé-Mengual et al. (under review). For this, we considered intensive use as the worst-case scenario, although this might not apply to all taxa and impact categories (e.g., ecosystem services) because of the wide range of responses. An alternative would have been to assume a representative case based on, e.g., national statistics. However, we have chosen the more conservative approach of assuming an intensive land use to help incentivize LCI database developers to be specific, as previously recommended by Koellner et al. (2013a). Following a similar logic, for unspecified flows (where even the land use type was unspecified, and not only the intensity level), we attributed the flow to the category "intense urban land use." As an exception to the latter rule, we assumed the flow "unspecified natural (non-use)" as pasture with a minimal use intensity, as only grasslands (under pasture) and forests (under managed forests) can be natural and no use suggests a minimal intensity if it is still occupied. Theoretically, natural land cover without any use would have zero impact and should not be assigned to any characterization factor; however, existing occupation flows contradict the assumption that there is really no use. In turn, there is a risk

[1] Also representing elementary flows in the World Food Database.

that this underestimates the transformation impact if an area actually is transformed from a natural (non-use) state.

Where the land use type is indicated as arable land or agriculture, it could represent either annual or permanent crops or, for agriculture, even pasture. In such a case, we assumed it to represent annual crops for two reasons: (1) Koellner et al. (2013a) subdivide agriculture into arable and permanent crops, while pasture is part of a different parent category, and they describe arable land use as annual crop production, and (2) it again represents a more conservative approach, as the characterization factors for annual crops reflect, on average, larger impacts than those for permanent crops or pasture (Chaudhary et al. 2016). Likewise, EXIOBASE sometimes mixes annual and permanent crops within the same product category, e.g., “Vegetables, fruits, nuts,” and we then assigned it to annual crops for the LCIA method.

3 Results and discussion

LCI databases cover much more land use classes than LCIA methods, although not all elementary flows are linked to any unit processes. The development of LCIA methods requires many data, and it is, thus, challenging to refine the land use classes. It needs some flexibility and the setting of priorities.

All databases and nomenclature systems analyzed include land use classes with an uncertain matching to the LCIA methods (Table 1). This uncertainty applies mostly to unspecified intensity levels but sometimes also to the land use types. Not all such flows are linked to a process. However, for example, “unspecified” is linked to several processes in ecoinvent. A notable example where the matching is not obvious is “arable, greenhouse.” Although we usually linked arable land in LCI databases to annual crops in the LCIA methods, we linked “arable, greenhouse” to light urban land use. As proposed by Chaudhary and Brooks (2018), we linked a “mineral extraction site” to intense urban land use, as it is closer to urban land use than any of the other classes included in the LCIA methods and it falls into the same class of artificial areas as urban land use in Koellner et al. (2013a). Characterization factors for intense urban land use are likely to underestimate the impacts of a mineral extraction site, but not linking them to any characterization factor would imply that the impacts are completely ignored. Some LCI databases also cover numerous unit processes linked to mineral extraction sites (e.g., > 200 occupation flows in ecoinvent). So, it could be valuable for future LCIA methods to provide characterization factors specific to mineral extraction sites.

Plantations, which refer to timber plantations, are included in the LCIA method by Chaudhary and Brooks (2018) but could not be linked to any elementary flow in the LCI databases, the nomenclature, or the environmentally extended MRIO database. We linked all forests to the characterization factors for managed forests. The standardized land use classification by Koellner et al. (2013a) does also not distinguish between timber plantations and managed forests. Both Chaudhary and Brooks (2018) and Koellner et al. (2013a) consider other plantations, e.g., for oil palm, under (permanent) crops. Agroforestry where trees or shrubs are mixed with crops and/or pasture on the same field are encoded in some databases as “agriculture, mosaic” or “heterogeneous, agricultural.” They are also relevant as a land use class in LCIA methods because agroforestry can enhance biodiversity in an area compared to conventional agriculture or forestry (Torralba et al. 2016). In the standardized classification by Koellner et al. (2013a), “agriculture, mosaic” also forms its own class and is not a subcategory of any other class.

Some elementary flows remained unmatched (Table 1). These are related mostly to water, i.e., oceans (seabed and benthos), water bodies on land (rivers and lakes), and wetlands. These would be assessed through different impact categories than land use, which is relevant to terrestrial ecosystems. Especially for marine ecosystems, there is currently a lack of impact assessment methods (Woods et al. 2016). For artificial freshwater bodies, Sanyé-Mengual et al. (under review) took a different approach instead and decided to assign artificial water bodies the highest available land occupation CFs to adopt a precautionary approach. We suggest to apply the characterization factors by Dorber et al. (2020) for terrestrial biodiversity impacts from inundation for land aquaculture or reservoir construction. The authors distinguish one natural and four anthropogenic land use types before occupying the land with the artificial water body. However, they do not provide characterization factors for land transformation and did not examine potential benefits for aquatic species who gain habitat.

For land transformation flows, it seems helpful to include only the net inventories. Currently, LCI databases sometimes have the same land use class for transformation “from” and “to” flows of the same unit process, which might even have the same value. For example, “transformation, from annual crop” would be 1 m^2 and “transformation, to annual crop” would also be 1 m^2. It should not affect the final results, as characterization factors with an opposite sign are applied to both so that the impact cancels out, but it makes the results of flow contribution analyses hard to interpret and can be confusing, as it implies that no transformation took place (recently). While land occupation always follows land transformation, impacts from land transformation are not always considered. For example, Koellner et al. (2013b) suggest allocating land transformation impacts to the production output of the first 20 years, which is consistent with assessments of greenhouse gas (GHG) emissions driven by land use change. However,

Table 1 Summary of the matching exercise across the analyzed databases/nomenclatures

Database/nomencl	Number and typology of elementary flows	Coverage (matched vs. excluded flows)	Need for assumpt. (%)*	General comment
Agri-footprint ® v5.0	9 elementary flows in total, among which: • 3 "occupation" flows • 4 "transformation from" flows • 2 "transformation to" flows	9 (100% of the total) vs. 0	100%	Agri-footprint® is a life cycle inventory (LCI) database that has been developed by Block Consultants (the Netherlands) and released in 2014. It covers the agriculture and food sector and contains data on different types of agricultural products (feed, food, and biomass)
Agribalyse v3.0.1	145 elementary flows in total, among which: • 50 "occupation" flows • 46 "transformation from" flows • 49 "transformation to" flows	126 (~87% of the total) vs. 19	~37% (=46/126)	The Agribalyse database has been produced as part of the Agribalyse programme led by ADEME and INRAE (France) since 2009. It contains agricultural and food products produced and/or consumed in France. It follows CIQUAL nomenclature, the French nutritional database
Ecoinvent v3.4	177 elementary flows in total, among which: • 57 "occupation" flows • 60 "transformation from" flows • 60 "transformation to" flows	138 (~78% of the total) vs. 39	~43% (=60/138)	The ecoinvent database is produced by a not-for-profit association based in Switzerland and is dedicated to promoting and supporting the availability of environmental data worldwide. The database contains information on process data for thousands of products, going well beyond the agriculture and food sector
EF nomenclature v3.0	174 elementary flows in total, among which: • 58 "occupation" flows • 58 "transformation from" flows • 58 "transformation to" flows	156 (~90% of the total) vs. 18	~48% (=75/156)	The EF nomenclature is the nomenclature system adopted by the Environmental Footprint (EF) reference package, which is used in the development of EF compliant datasets (Fazio et al. 2019). The classification of land use flows is based on the classification proposed by Koellner et al. (2013a), as reported in Sala et al. (2019)
EXIOBASE v3.4	25 land use extensions, among which: • 25 (i.e. all) related to occupation	25 (100% of the total) vs 0	100%	EXIOBASE 3 is an environmentally extended multi-regional input–output database that covers 49 countries/regions and 200 product categories. It also includes environmental information on emissions, energy use, water use, material use, land use, and waste (Stadler et al. 2018)
GaBi	225 elementary flows in total, among which: • 75 "occupation" flows • 75 "transformation from" flows • 75 "transformation to" flows	51 (68% of the total) vs. 24 flows per type	~47% (=24/51)	The GaBi Database is maintained by thinkstep AG, a Sphera company (formerly known as PE INTERNATIONAL AG). Its development started over 30 years ago, and the database is updated annually. With over 15,000 plans and processes, it provides the largest industry coverage of LCI data worldwide
US LCI Database	106 elementary flows in total, among which: • 42 "occupation" flows • 32 "transformation from" flows •32 "transformation to" flows	95 (~90 of the total) vs. 11	~34% (=32/95)	The US LCI database has been developed under the lead of the National Renewable Energy Laboratory (NREL). The database has been publicly available since 2003 and covers materials, products, and processes commonly used in the US

*The % value is calculated from the ratio between the matched flows indicated in gray font color and the total number of matched flows (i.e., excluding the unmatched flows) the allocation period is arbitrary and some LCI databases, such as ecoinvent, implement a more flexible approach for land use change-related GHG emissions.

Although LCI databases provide information on transformation "from" flows, characterization factors are not specific to the land use class before any transformation. This is relevant to occupation and transformation impacts. Both types of impacts depend on the difference in biodiversity between the current land use and a reference situation (see Eqs. 1–4 in Koellner et al. 2013b). The reference situation in the LCIA methods by Chaudhary and colleagues (2016, 2018) is a mix of (semi-)natural land use or even all species in the entire ecoregion, which might overestimate the species richness of the reference land use if the ecoregion consists of multiple natural habitat types with different species compositions. We recommend that LCIA method developers put effort into making characterization factors specific to the natural habitat before the transformation, as, for example, Scherer et al. (2020) did. Scherer et al. (2020) found in a German case study that occupation with agricultural land causes higher functional plant diversity loss if it replaces broad-leaved forest than if it replaces coniferous forest. Such differences might be more pronounced if we distinguish, for example, forest and grassland as the land use before occupation, since they provide habitat for different species with different affinities for agricultural land. Where the land use before the transformation is also anthropogenic, the difference between characterization factors can be used. However, making the characterization factors specific to the land use before transformation might imply a need for changes in the structure of LCI databases and the design of LCA software.

Available LCI databases do not yet consider different landscape patterns, such as land fragmentation. The LCIA method by Kuipers et al. (2021) assesses the impacts of land fragmentation at the level of terrestrial ecoregions and is designed such that it does not require additional information from LCIs. However, it would still be useful to collect such information in LCIs, as there could be considerable differences within large ecoregions.

Besides our work and the work by Sanyé-Mengual et al. (under review), there are other ongoing efforts to improve the connection between inventories and impact assessment methods more generally. The task force on cross-cutting issues within the Life Cycle Initiative includes a subtask on the LCI-LCIA connection with a much broader scope. Within the last phase of GLAM, they identified 15 key issues grouped into four domains, which hinder such a connection. We addressed here some of those issues but limited to the land use elementary flows only. Currently, that subtask works on establishing an open-access and traceable system together with recommendations and mitigation measures for a coherent connection between LCI and LCIA in close collaboration with key stakeholders such as database and software providers (R. Hischier, personal communication, September 2021).

4 Conclusions

Inventory analyses and the development of global LCIA methods can both require immense efforts. Understandably, it is not always feasible to provide all the details that may be desirable. Based on our observations during the matching of land use classes between LCI databases and LCIA methods, we provide some recommendations, which may help set priorities on improving further developments. Note that this is not an exhaustive list and there are several other opportunities for improvement, some of which we pointed out above.

For LCI database developers, we recommend to:

- make the inventory more specific wherever possible and avoid unspecified land use types or intensities, which are now penalized with assuming the worst case
- do not offer occupation flows for natural land covers that are supposed to be without any use
- include only the net inventories for land transformation
- distinguish managed forests and timber plantations, as done in some LCIA methods

For LCIA method developers, we recommend to:

- provide a comprehensive matching to the most important LCI databases, similar to what we have done here
- distinguish annual and permanent crops

5 Data

An Excel file accompanies this article and contains our proposed matching of land use classes between LCI databases (and one nomenclature and one MRIO database) and LCIA methods. Each database is presented in a separate sheet.

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Declarations

Conflict of interest The authors declare no competing interests.

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