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National inventories of land occupation and transformation flows in the world for land use impact assessment

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Abstract

Purpose: Land use can cause significant impacts on ecosystems and natural resources. To assess these impacts using life cycle assessment (LCA) and ensure adequate policy-making, comprehensive national inventories of land occupation and transformation flows are required. Here, we aim at developing globally-differentiated inventories of land use flows that can be used for primary use in life cycle impact assessment or national land planning.

Methods: Using publicly-available data and inventory techniques, national inventories for several land use classes were developed. All land use classes were covered with the highest retrievable level of disaggregation within urban, forestry, agriculture, and other land use classes, thus differentiating 21 land use classes. For illustrating the application of this newly-developed inventory, two different application settings relevant to life cycle impact assessment were

considered: the calculation of global normalisation references for 11 land use impact indicators related to soil quality assessment (adopting the methods recommended by the EU Commission) and the determination of generic globally-applicable characterisation factors (CFs) resulting from aggregation of country-level CFs for situations for use when land use location is unknown.

Results and discussion: We built national inventories of 21 land occupation and 17 land transformation flows for 225 countries in the world for the reference year 2010. Cross-comparisons with existing inventories of narrower scopes attested its consistency. Detailed analyses of the calculated global normalisation references for the 11 land use impact categories showed different patterns across the land use impact indicators for each country, thus raising attention on key land use impacts specific to each country. Furthermore, the upscaling of country-level CFs to global generic CFs using the land use inventory revealed discrepancies with other alternative approaches using land use data at different resolutions.

Conclusions and recommendations: In this study, we made a first attempt at developing national inventories of land use flows with sufficient disaggregation level to enable the calculation of normalisation references and differentiated impacts. However, the findings also demonstrated the need to refine the consistency of the inventory, particularly in the combination of land cover and land use data, which should be harmonised in future studies, and to expand it with differentiated coverage of more land use flows relevant to impact assessment.

Keywords

LCA, life cycle impact assessment, LANCA, spatial differentiation, normalisation reference, urban area, soil quality indicator, life cycle analysis.

1. Introduction

Life cycle assessment (LCA) is a decision-support methodology used to quantitatively assess the environmental impacts associated with a product or system's life cycle, i.e. from the extraction of the raw materials through the production and use or operation of the system up to its final disposal (ISO 2006). In the assessment, emissions of pollutants and resource consumption are compiled throughout the entire system life cycle and translated into impact indicator scores with the use of specific characterisation factors in the life cycle impact assessment (LCIA) phase (Hauschild 2005). As an optional step in LCIA, normalisation of the impact indicator results can be performed to facilitate interpretation and communication of the results (ISO 2006). This step consists in dividing the impact results associated with the system under study by normalisation references, which are impacts relating to a reference situation, typically the background load of a region (e.g. Europe) or the entire world (Laurent and Hauschild 2015).

With respect to land use impact assessment, several LCIA methods have emerged to enable the quantification of impacts of land occupation and land transformation on biodiversity (e.g. De Baan et al. 2013, 2015; Teixeira et al. 2016), biotic production (e.g. Bos et al. 2016; Brandão and Milà i Canals 2013; Milà i Canals et al. 2007; Taelman et al. 2016), etc. Some of these methods address spatial differentiation and thus provide characterisation factors (CFs) at grid cell level or at country level (e.g. Bos et al. 2016; Chaudhary et al. 2015, 2016; Chaudhary and Brooks, 2018; Morais et al. 2016). To provide generic CFs when the exact location of the land use is unknown, upscaling of the CFs across countries or grid cells is necessary. Regionalised information about areas of specific occupied or transformed land types may, therefore, be needed when these are used as weighting proxies for upscaling the CFs from fine spatial resolution to regional, continental or global scale (Verones et al. 2017). In addition, to enable the consistent use of normalisation in LCA studies, normalisation references should be available for all impact categories, including land use. For their calculations, country-specific inventories of land use types that match the level of spatial differentiation of the LCIA methods must be built.

Until now, to the authors' knowledge, only one publicly available inventory of land use flows used in the determination of normalisation references exists at a global scale, i.e. the inventory built for the year 2000 by Wegener Sleeswijk et al. (2008). However, that inventory does not differentiate between countries, and only accounts for two land use flows, i.e. "agricultural land occupation" and "urban land occupation", hence not addressing land transformation. Therefore, there is a current lack of national land use inventories, which allow consistent coupling with newly developed spatially-differentiated LCIA methods.

In this study, we thus aim at bridging this gap and (1) develop an up-to-date global country-specific inventory of land occupation and transformation flows that can be used in LCIA for assessing land use impacts; (2) apply this inventory to illustrate its use and benefits in LCIA method development. Two application cases are considered: determining normalisation references for selected LCIA methods and exploring the use of the developed inventories for upscaling of CFs. The retained LCIA methods are the spatially-differentiated LANCA LCIA method (Bos et al. 2016) and the method by Milà i Canals et al. (2007). This selection was motivated by the fact that LANCA offers a case of spatially-differentiated method application (selected in an updated form for product environmental footprint by the EU Commission; Sala et al. 2016; De Laurentiis et al. 2018; Crenna et al. 2018). The method by Milà i Canals (2007), which is also part of the EU recommended LCIA practice (EC, 2011; Hauschild et al, 2013), enables comparison with previously assessed impacts (i.e. normalisation references computed by Sala et al. 2015), hence its additional selection.

2. Materials and Methods

2.1. Overall methodology

To accommodate the different sourcing and further processing of the data, the inventory methodology addressed four distinct groups of land use, namely agriculture, forestry, urban and other land use classes. The outputs from each were gathered to form the final inventory. Data for

these four groups of land use flows were hereafter combined with country-specific characterization factors for each of the two LCIA methods to calculate global normalisation references and support further analysis (see Figure 1). The subsequent sections detail the materials and methods used for the inventory building (Sections 2.2 and 2.3) and the impact assessment (Section 2.4).

<Figure 1>

2.2. Inventory building

2.2.1. Land use flow coverage

In our land use inventory, we covered four major groups of land use classes: urban, agriculture, forestry and other land use classes (see Table 1). Each main group was further divided into sub-groups of land use classes. The latter group includes seven classes of land cover: shrub land, bare soil, herbaceous vegetation (aquatic or regularly flooded), sparse vegetation, mangroves, water bodies and snow/glaciers. All sources are deemed sufficiently up-to-date to allow building comprehensive inventories for 2010. The year 2010 was found to be associated with the best consistency and comprehensiveness with respect to temporal and geographical scopes of the data available from the different sources (see Table 1). It was thus selected as the reference year for the inventory and the corresponding normalisation references. The data sources mainly consist of globally-recognised and authoritative institutions, e.g. the Food and Agriculture Organisation of the United Nations (FAO) and the National Aeronautics and Space Administration (NASA).

For each of the land use flows, occupation and transformation data for the year 2010 were estimated at a country-scale resolution. The land occupation data describe the time-integrated area of a given type of land in the reference year and is expressed as ha.yr. The land transformation data describe the area change of a given land use type between two reference points in time, typically between two consecutive years. It is expressed in ha, and, unlike occupation data, it can thus be negative (if some areas of a given land use type are converted to another land use type, thus resulting in a loss

of the original land use type). As a result, transformation flows in the final national inventories may be negative.

Occupation data were primarily compiled from different data sources and complemented with further data processing (see Table 1). Following the study made by Sala et al. (2014), occupation and transformation data from the United Nations Framework Convention on Climate Change (UNFCCC) were disregarded as they were reported to be associated with inconsistencies and a lack of completeness to meet the objectives of the present study. UNFCCC thus does not provide its National Inventory Reports for all the countries in the world, and countries such as China, India, Brazil, and African countries are not included (UNFCCC 2013). In addition, only five land use classes are distinguished in the UNFCCC reports, thus limiting the objective of a differentiated inventory.

While some global land transformation data exist, in particular from climate research studies modelling land use – climate interactions (e.g. Hurtt et al. 2011), they have been reported to embed large inconsistencies in their present-day land use estimates because of yet-unresolved inconsistencies between land cover and land use definitions associated data incompatibility (Prestele et al. 2017; see discussion in Section 3.4). Therefore, to remain consistent with our occupation data, we opted for the simplified approach to derive transformation data as differences of derived occupation data between years. Detailed methodologies for the collection and handling of occupation and transformation data for each land use class are documented in the following subsections. These methodologies were applied to obtain consistent national inventories. Available inventories only covering European countries (e.g. Sala et al. 2014) were disregarded to maintain consistency between countries in the world. However, those were considered for the validation of our inventory –see Section 2.3.

<Table 1>

2.2.2. Inventory of urban flows

To the authors' knowledge, no up-to-date, reliable estimates of urban land use flows at the global scale are available. At such a scale, the Lincoln Institute of Land Policy (2016) reported country-specific urban land cover estimates for the year 2000. In a parallel work, Natural Earth (2016) provided a shapefile of urban areas as areas of dense human habitations (without indicating the considered urban population density threshold) derived from 2002-2003 MODIS satellite data at a 1-km grid resolution (Schneider et al. 2003).

In our study, a national inventory was generated by extracting data from the mapping (raster format) made by Hoskins et al. (2016) using QGIS software (<http://www.qgis.org/en/site/>). The mapping of Hoskins et al. (2016) was considered the most up-to-date and consistent land cover inventory at the time of the study. It includes harmonised and validated data for five land use classes for the year 2005, including urban land. The 2005 urban occupation data from Hoskins et al. (2016) were assumed representative for the year 2010 and were used as such in the current study. This assumption was tested by investigating the influence of urbanisation between 2005 and 2010. To do so, a method was developed based on the use of population density as an indicator for urban areas. In the absence of any other means to derive transformation data, this method was also used to estimate the urban land transformation flows for the different countries, as described below.

Data on population density for 2010 were extracted from NASA (2016), taking the recommended set of adjusted data for regional and global analysis (CIESIN 2016). The data were available in raster format at a spatial resolution of 30 arc-second² (ca. 1 km²) and were processed using QGIS software. A threshold for the urban population density was set up to characterise urban areas. That threshold value was set to 300 inhabitants/km² following the definition of urban areas provided in the OECD methodology (EUROSTAT 2016). The OECD methodology supplemented the use of the thresholds with a requirement for the adjacency of several urban-identified grid cells (EUROSTAT

2016). However, this second criterion was not considered in the current study because of the relatively large area covered by a single grid cell (ca. 1 km²) and the objective to map all urban areas including small cities.

The urban land occupation flows were estimated for each country by computing the number of grid cells above the population density threshold and assuming the area reported in QGIS for each grid cell (i.e. pixel area of ca. 0.86 km²) in lieu of the approximated 1 km² described in CIESIN (2016). The urban transformation flows were derived by using occupation data at the years 2005 and 2010 (i.e. the two years available in the NASA population density grids). The differences of occupation data between these two years, divided by a factor of 5, were assumed to provide the average transformation flows for 2010, thereby reducing the risk of integrating effects of large fluctuations in land transformation between two specific years. This approach was assumed consistent with the occupation inventory extracted from data in Hoskins et al. (2016).

2.2.3. Inventory of agricultural flows

Occupation data for each agricultural land use class (see Table 1) were extracted from the statistics provided by FAOSTAT (2016), which releases times series at country scale resolution. These statistics originate from official national reports, which are aligned with FAOSTAT land use classification. With the exception of obvious inconsistencies, these reports are not corrected by FAO, thus resulting in possibly large differences in data quality across countries (FAOSTAT 2016; Hansen et al. 2013; see also discussion on uncertainties in Section 3.4). To report occupation flows, the FAOSTAT dataset for the year 2010 was used for building the inventory.

It is noteworthy that, in the main group “agriculture”, the agricultural areas (i.e. sub-group 2) represent the total areas of other specific flows such as arable land (nr. 2.1 in Table 1), permanent crops (nr. 2.2), and permanent meadows and pastures (nr. 2.3). Likewise, the sub-group “arable land” (nr. 2.1) is the aggregation of fallow land (temporary), temporary crops, and temporal

meadows and pastures (nr. 2.1.1-2.1.3). In the calculation of the normalisation references, double counting can thus be avoided by dismissing the total aggregated flows, which are already included through more specific flows. For example, in the LANCA LCIA method, “fallow land (temporary)” was used to match the CF available for the flow “fallow land” and the total arable land excluding the fallow land was used to match the CF for the flow “arable land”.

Transformation data were estimated by extracting occupation data for all agricultural land use flows for each year in the period 2005-2011 from FAOSTAT (2016). Transformation flows were estimated as the differences between two consecutive years of occupation data, e.g. the transformation flows for 2010 being the difference of occupation data between 2011 and 2010. Similarly to the approach for deriving urban transformation flows, to avoid integrating effects of large fluctuations in land transformation between two specific years, the transformation flows for 2010 were calculated by taking the average of each of the transformation flows over the period 2005-2010.

2.2.4. Inventory of forestry flows

As for the agricultural classes, statistics from FAOSTAT (2016) database were used for forestry flows. Occupation data for each country in the year 2010 were extracted for four flows: “forest”, “primary forest”, “other naturally regenerated forest”, and “planted forest”. The class “forest” represents the total area of the other three classes. The total area of planted forests was further differentiated into extensive and intensive forest areas, using data from FAO (2010).

Transformation flows were estimated in the same way as agricultural land transformation flows, taking the average of the transformation flows calculated for each year over the period 2005-2010 using occupation data (see Section 2.2.3).

2.2.5. Inventory of other land use classes

Occupation data for shrub land, bare soil, herbaceous vegetation (aquatic or regularly flooded), sparse vegetation, mangroves, water bodies and snow/glaciers were retrieved from the maps of “dominant land cover types” provided by FAO (2014). The original dataset was created by data fusion of different sources with different reference years, e.g. CORINE land cover data for Europe (Latham et al. 2014). However, given that the data lie within the period 2000-2010 for most countries, the estimates are expected to be representative of the year 2010 at country and global levels. The data being provided with a spatial resolution of 30 arc-second² (ca. 1 km²), the same computing approach as for urban occupation flows, i.e. aggregation of pixels in QGIS software, was used (see Section 2.2.2).

No datasets providing time series of occupation data for the classes of shrub land, bare soil, herbaceous vegetation (aquatic or regularly flooded), sparse vegetation, mangroves, water bodies and snow/glaciers could be retrieved. Differentiated transformation data for these flows could therefore not be calculated and instead, a grand total transformation flow was derived by assigning to these grouped land classes the opposite value of the sum of transformation flows for all other classes (to get the total transformation flows equal to zero for each country).

2.3. Validation of the inventory with different data sources

To evaluate the precision and accuracy of the inventory, comparisons with other inventories and data sources were performed. With only a few exceptions, like the total forestry flows provided in Hansen et al. (2013), the absence of inventories addressing land transformation data led to direct these comparisons to only addressing occupation data. Since transformation data are derived from occupation data, this is considered an acceptable testing approach for the entire inventory.

Urban occupation areas, which were estimated in the current study, were compared with estimates from four alternative data sources and previous works: (i) the global inventory from Atlas of Urban

Expansion (AUE) at year 2000 (Lincoln Institute of Land Policy, 2016); (ii) MODIS global inventory at year 2002-2003 (Natural Earth, 2016); (iii) CORINE Land Cover European inventory at year 2006 (EEA 2014); and (iv) EU-27 inventory at year 2010 by Sala et al. (2014). It was additionally compared to the occupation values obtained for 2005 and 2010 using the urban population density approach, as described in Section 2.2.2.

For the occupation flows “arable” and “forest”, the current inventory was compared with the EU-27 inventory reported by Sala et al. (2014). “Cropland” and “pasture” occupation flows were also compared with those reported in Hoskins et al. (2016). Furthermore, both occupation and transformation data for total forestry were compared with the forest cover mapping performed by Hansen et al. (2013). For the flow “water bodies”, comparisons were performed with FAOSTAT data, which were hypothesised to be less complete due to a definition of inland water limited to major rivers, lakes, and reservoirs.

To validate the complete inventory, the total land use area estimated from our inventory per country was calculated and compared with the total area of each country as reported by other sources. Such aggregated data are more accurate and accessible via public databases. For this comparison, data from FAOSTAT (2016) that covered 191 countries belonging to our derived inventory were used. A final comparison was also performed at the level of the normalisation references (see Section 2.4.2) using the normalisation references calculated for urban and agricultural land occupation by Wegener Sleeswijk et al. (2008) for the year 2000.

2.4. Life Cycle Impact Assessment

2.4.1. LCIA methods for land use

The derived global inventory of land use flows allows calculation of normalisation references for land use for any LCIA method that includes this impact category. Here, only two LCIA methods were considered as an illustration of the inventory application, i.e. the method by Milà i Canals et

al. (2007) and the LANCA method v. 2.2 (Bos et al. 2016), which corresponds to an updated version of the model developed by Baitz (2002) and Bos and Wittstock (2007). The method by Milà i Canals et al. (2007) is part of the International Reference of Life Cycle Data System (ILCD) recommended sets of LCIA methods (EC-JRC 2011; Hauschild et al. 2013), whereas the LANCA method has been selected for the environmental footprint method (Sala et al. 2016; Crenna et al., 2018).

The method derived by Milà i Canals et al. (2007) has a focus on soil quality, and its indicator describes the changes in soil organic matter (SOM) associated with land interventions. Indicator results are thus expressed as kg-C, reflecting changes in soil organic carbon (Milà i Canals et al. 2007; EC 2011). The method provides generic, i.e. non-spatially-differentiated, CFs. The updated method by Brandão and Milà i Canals (2013) may be considered more appropriate as it has a global scope and a classification by climate regions. However, based on the CF values reported in Brandão and Milà i Canals (2013) and Milà i Canals et al. (2013), it can be observed that differences between climate regions are relatively minor compared to those between land classes taken at a detailed level (e.g. irrigated arable land vs. non-irrigated arable land) and that significant land use classes (e.g. forest) have no characterisation factor. Because such level of details in the land classes could not be achieved (see Section 2.2), the method by Milà i Canals et al. (2007) was therefore retained. This also enabled to keep the objectives of deriving normalisation references for the ILCD recommended LCIA methods; see above.

In contrast to the above method, the LANCA method includes country-specific CFs covering 212 countries, and addressing five distinct soil quality-related impact categories: (i) erosion resistance (ii) mechanical filtration (iii) physicochemical filtration (iv) groundwater regeneration, and (v) biotic production. For each indicator, different CFs addressing occupation and transformation flows respectively are provided (De Laurentiis et al., 2018; Crenna et al., 2018). Therefore, the LANCA method encompasses ten impact categories (five addressing occupation and five addressing transformation flows). The provision of normalisation references for the LANCA method may help

in the reduction and/or aggregation of these indicators, as was highlighted in the review of LCIA methods reported in EC (2011), where the scientific soundness of the LANCA method was praised but its applicability with too many indicators forbade its recommendation (see De Laurentiis et al., 2018, exploring this topic).

2.4.2. Calculation of normalisation references

The normalisation references were calculated for each of the selected LCIA methods by multiplying the generic or country-specific CFs with the global or the country-specific land use flows from the developed inventory. The aggregation of the impacts across all countries, divided by the world population in 2010, led to determining the global per capita normalisation references.

It is noteworthy that to characterise the transformation flows aggregating the seven land use classes of shrub land, bare soil, herbaceous vegetation (aquatic or regularly flooded), sparse vegetation, mangroves, water bodies and snow/glaciers (see Section 2.2.5), a weighted average of the CFs of all seven flows was calculated based on their occupation flow distribution in each country (i.e. using the area of each land class in the country as weighting factor). CFs for “transformation to” flows were considered in this exercise. The resulting country-specific CFs were applied to the transformation flow inventory data in the calculation of the normalisation references. This approach comes to assume that the relative transformation in these seven classes is evenly distributed within each country; this assumption – difficult to avoid in the current inventory – is however recognised as debatable, and should be refined in future works.

2.4.3. Upscaling of the spatially-differentiated characterisation factors

In LCA case studies, for which the precise location or country, in which the land use takes place, is unknown, generic characterisation factors are required. To maintain consistency in the assessment

results, these generic CFs need to be spatially-representative and capture the geographical variations of the CF, typically using weighting by a relevant parameter to upscale the spatially-differentiated CFs into a generic value (Verones et al. 2017). In this study, we investigate the influence of different upscaling techniques on the resulting generic CF values, taking the LANCA method as an example (the method by Milà i Canals does not include spatial differentiation).

In the LANCA method, Bos et al. (2016) defined average CF values using weighting by total surface areas. As an alternative upscaling technique, we used the developed national inventories of land occupation or transformation flows, which were taken as land-class-specific, country-specific weighting proxies. Such an approach was also applied in previous studies with smaller scopes like in Morais et al. (2016) for Europe. These thus-defined proxies were combined with corresponding country-specific CFs and aggregated into generic CFs for each land occupation or transformation flows covered in the inventory –see Equation 1.

$$CF_{generic}^j = \frac{\sum_i A_{occ,i}^j \times CF_i^j}{\sum_i A_{occ,i}^j} \quad (\text{Equation 1})$$

With $A_{occ,i}^j$ the area (or area occupied over time for occupation flows) for land occupation or transformation flow i in country j , and $CF_{generic}^j$ and CF_i^j the generic and country-specific characterisation factors for land use occupation or transformation flow j (e.g. primary forest or urban areas).

3. Results and discussion

3.1. National inventories of land use occupation and transformation flows for 2010

The resulting inventory overall covers 225 countries and territories, 23 occupation flows and 17 transformation flows, including five totals for forest, planted forest, agriculture, arable land, and

arable land without fallow land. The transformation flows include the same land use classes as occupation flows, minus shrub land, bare soil, herbaceous vegetation (aquatic or regularly flooded), sparse vegetation, mangroves, water bodies and snow/glaciers, for which a lumped transformation flow was derived (see Section 2.2.5). The complete inventory, differentiated by countries and land use flows, is available in Table S1 in Electronic Supplementary Materials.

All major flows typically covered in LCIA methods for land use impact assessment are expected to be covered, although the CFs provided in some LCIA methods typically have a finer differentiation of land classes, e.g. irrigated vs. non-irrigated land flows in LANCA method. Such level of details could not be retrieved for all countries at a global scale. Overall, only a few countries are missing due to unavailability of data, although the lack of data increases dramatically the finer the land class differentiation is (e.g. 151 countries with “NA” for temporary crops; see Table S1).

Aggregating the country-specific total land area wherever the land use flow is covered and comparing it to the global total land area, it is observed that all land use flows are mapped with a global coverage above 99.1% except for permanent crops (95%) as well as fallow land (i.e. 23%), temporary crops (37%) and temporary meadows and pastures (8%), all three latter being further specifications of arable land (total with a coverage of 100%). Overall, such coverage demonstrates an acceptable global representativeness of the inventory, which implies a sufficient level of land-class and country differentiation to enable the calculation of consistent normalisation references for the land use impact category at national, regional and global scales for any existing LCIA methodologies.

3.2. Validation of the developed inventory

3.2.1. Urban areas

<Figure 2>

Figure 2 shows the comparisons of country-specific urban land areas between available data sources (global and European levels). R-squared values (R^2) above 0.72 and 0.80 can be observed for global and European datasets. However, some discrepancies may occur at a country level, with factors up to one order of magnitude difference between available sources (see black dots and red triangles in Figure 2) and with even larger differences when comparing to results from the urban population density approach tested in this study (blue triangles in Figure 2). Three main causes may explain this trend: (i) the inventorying methodology applied, (ii) the considered definition of urban areas, (iii) the reference year of the inventory.

The method applied to identify and extract urban areas may differ from one source to the other. For example, in the extreme case of the urban population density approach (blue triangles in Figure 2), a raster approach was adopted, implying the counting of pixels with a population density above 300 inhabitants/km². A shortcoming of this approach is that grid cells, which meet the above population density criteria but comprise several land uses (e.g. a combination of urban land with agricultural land or water bodies), end up fully assigned to urban land. It thus generates overestimations in the occupied urban land area and overlaps with other types of land use classes (e.g. agricultural land or water bodies). On the other hand, urban land areas may also be underestimated when areas of land that are dedicated to constructions and urban building structures do not exceed the threshold of 300 inhabitants/km² and are thus not accounted for in the total urban land areas. The resulting influence of these two counteracting uncertainties on the overall uncertainty of the estimates for urban land and the implications for the entire land use flow inventory are difficult to assess, although these are believed to be acceptable given the objective of the study to compile macro-scale (national, regional and global levels) aggregates capturing average land use conditions. In the context of deriving transformation flows, some of those uncertainties, if present in both sets of 2005 and 2010 data, may cancel each other out. Hence, it was deemed acceptable in the current study, although remaining a definite source of improvements for further studies. Potential ways to correct for these systematic overestimations or underestimations underlying in this inventory approach, e.g. by

relying on more comprehensive and highly-resolved maps below 1-km² grid cells, should be explored in future inventorying works.

The definition of urban areas may also significantly contribute to the discrepancies observed. Each data source relies on their own definitions of urban area, which can vary considerably in the scoping of what is included or excluded (e.g. artificial areas, continuously built-up areas, etc.). Differences between the reference years of the different data sources may also partly explain the different results obtained for countries, which have known rapid urbanisation changes in the period 2000-2010 (UN 2014).

Finally, another comparison can also be made with the inventory calculated by Wegener Sleeswijk et al. (2008), who derived ReCiPe 2008 normalisation references for urban land occupation in 2000 of 1.89E+11 m².yr and 4.71E+12 m².yr for EU25+3 and the world, respectively. These normalisation references are directly comparable to the inventory made in this study as ReCiPe 2008 LCIA method for land use assumes land competitiveness and sets all CF to 1 (Goedkoop et al. 2009). The total values obtained for EU25+3 and the world in the current inventory are equal to 1.32E+11 m².yr and 6.95E+11 m².yr, which are lower by factors of 1.43 and 6.8 than the values in Wegener Sleeswijk et al. (2008). The differences for urban areas in EU25+3 are likely to be explained by different urban areas definitions (see above), but seem overall consistent with other estimates of 1.77E+11 m².yr (retrieved from the CORINE Land Cover 2006; EEA 2014) or 2.43E+11 (retrieved from 2010 inventory in Sala et al. 2014). At global level, the estimate in ReCiPe 2008 seems overestimated, when comparing to the aggregated global area from the current study and those from other studies, i.e. 6.06E+11 m².yr (Lincoln Institute of Land Policy, 2016) and 1.12E+12 m².yr (Natural Earth, 2016). The value derived in the current study falls within these two latter estimates, which renders it more plausible.

3.2.2. Cropland, pasture and forestry areas

<Figure 3>

Figure 3a-e shows the correlation between the estimates of the current study and those of other data sources for forest area ($r^2 = 0.84$ and 0.95 for global and EU scales), pasture ($r^2=0.94$ at global scale) and cropland/arable land ($r^2 = 0.95$ at both global and EU scales). It is noteworthy that in spite of relatively good correlation (r^2 above 0.8), the data in Hansen et al. (2013) end up systematically higher than the aggregated forest area totals derived from FAOSTAT data. This is a typical example of discrepancy generated when using either land use data or land cover data. While Hansen et al. (2003) relied on Earth observation satellite data, FAOSTAT data rely on a bottom-up collection of national reports about land use. Albeit being associated with uncertainties (e.g. inconsistent reporting across countries; Hansen et al. 2013), the latter data thus enable to better differentiate the extent of human activities on the land (e.g. extensive vs. intensive agriculture). In contrast, the tree cover data would include any tree cover irrespective of the type of land management and may thus encompass areas of plantation trees falling under cropland in FAOSTAT because of their food provision services (see also Section 3.4).

Like for urban land, an additional comparison can be performed using the 2000 normalisation references calculated for agricultural land occupation by Wegener Sleeswijk et al. (2008), who reported values equal to $2.10\text{E}+12$ and $3.30\text{E}+13 \text{ m}^2\cdot\text{yr}$ for EU25+3 and the world, respectively. In our current inventory, the total values obtained for EU25+3 and the world are found to be $1.88\text{E}+12$ and $4.91\text{E}+13 \text{ m}^2\cdot\text{yr}$, thus differing from Wegener Sleeswijk et al.'s values by factors of 0.90 and 1.49 , respectively. Considering the time lag of 10 years between the reference years of the two inventories, these values are believed to be consistent, thus suggesting a reasonable precision in the developed inventory.

3.2.3. Check of total land area per country

Figure 4 illustrates comparisons between the between the total land area calculated from the current inventory (i.e. sum of forestry, agricultural land, urban land, etc.) and the individual country areas reported in FAOSTAT (2018) and in the CIA World Factbook (CIA 2017). As reflected by the correlations ($r^2 = 0.90$ and 0.92 , respectively), an overall good agreement is obtained for most countries. However, as noted on the figure (i.e. spread of dots around the straight line $y=x$ in Figure 4), some countries show discrepancies: approximately 42 out of 225 countries/territories display differences in their total areas larger than 50% with at least one of the two alternative datasets. However, more than half are islands or small territories (e.g. Martinique, Saint Helena, etc.) as observed in Figure 4, which tends to show more discrepancies in the low end of the total areas than the high end. Table S3 reports the list of all 42 countries with the largest differences ($\pm 50\%$). The uncertainties in the computation method for some flows (map extraction) and the possible overlaps between land classes due to the mixing of land cover and land use data are likely to explain these differences –see Section 3.4. These also explain the globally aggregated total areas amounting to $1.64\text{E}+14 \text{ m}^2\cdot\text{yr}$ in this study, which is ca. 20% higher than the global estimates of ca. $1.36\text{E}+14 \text{ m}^2\cdot\text{yr}$ reported by FAOSTAT (2018) or CIA (2017).

3.3. Application of the globally-differentiated inventory

3.3.1. Calculation of normalisation references

Table 2 shows the normalisation references for the world for the recommended ILCD-LCIA method by Milà i Canals et al. (2007) and for the ten LANCA impact indicators. Out of the 212 countries and territories covered by the LANCA method, 199 were matched with data from our inventory. The remaining countries being small islands or territories (e.g. Gibraltar, Madeira, Saint Eustatius, Saint Maarten, etc.), their absences bear no influence on the calculation of the global normalisation references for the LANCA method.

To the authors' knowledge, no normalisation reference has ever been calculated for the LANCA method; hence, no comparison could be performed with alternative sources. With respect to the method by Milà i Canals et al. (2007), Sala et al. (2015) calculated a normalisation reference for EU-27 in 2010 equal to $7.58\text{E}+04$ kg-C/person/yr. The corresponding normalisation reference from the present inventory for EU-27 was found to be $5.65\text{E}+04$ kg-C/person/yr, thus decreasing by ca. 25% compared to the earlier inventory.

A number of uncertainties reside in the developed global land use inventory, in particular for specific countries and territories, which have been found to contain overestimations (see Section 3.4). However, given the geographical distribution of the global impacts between countries and the following identification of most contributing countries (see Section 3.3.2), these uncertainties are deemed to have a relatively minor effect on the global normalisation references. Caution should, however, be exercised for interpreting normalisation references for transformation-related impact categories, owing to the large uncertainties in the inventory of land transformation flows (see Section 3.4). Finally, it should be noted that a detailed analysis of the global normalisation references with contribution analyses can enable to identify land use flows causing the largest impacts at global and national levels (see Laurent et al. (2011) and Crenna et al. (2018) for such uses in LCIA). This information may thus serve as support for prioritising refinements in the current inventory or in land use LCIA method development.

<Table 2>

3.3.2. Geographical variations of land use impacts

At global scale, the countries with the largest contributions to global land use impacts (i.e. global normalisation references) are logically large countries, like Brazil, China, India, Indonesia, Australia, the United States, Russia and Canada (with, to a lesser extent, Mongolia, Pakistan and

Columbia for specific impact categories). All have contributions below/above $\pm 5\%$ global impacts in several impact categories.

To investigate national variations, national impacts normalised by total country areas were calculated –see illustrative examples in Figure 5. It can be observed that the geographical patterns change considerably from one impact category to another, thus reflecting that the relative magnitude of national land use impacts is strongly dependent on the land specificities of each country (types of land class represented, types of land use, soil properties, etc.). A further analysis should use these findings as a starting point and focus on assessing the importance of each impact indicator relative to the others to help policy-makers in prioritising the reduction of specifically high land use impacts. This can also be partly addressed through the aggregation of the different land use indicators, e.g. De Laurentiis et al. (2018) for the LANCA method.

<Figure 5>

3.3.3. Upscaling from country-specific to global characterisation factors

Following the upscaling technique defined in Section 2.4.3, a set of generic, globally-applicable CFs was calculated. These are documented in Table S2 (for the land use flows covered in the current inventory) along with the generic CFs derived by the LANCA method developers (Bos et al. 2016). Comparisons of the two sets of CFs show large differences across land use classes and impact categories. The two sets are found similar, i.e. with differences of less than 50% for most flows, for both mechanical and physicochemical filtration potentials. All CFs for urban land are also found close to each other. In contrast, for the remaining three impact categories, discrepancies and sign inversions are observed between the two sets of CFs. CFs for land classes, such as permanent crops, bare area or arable land thus differ by factors up to ca. 10 for some impact categories. CFs for forestry-related flows and pasture/meadows additionally tend to show sign inversions (e.g.

recalculated average CF value of $2.27\text{E-}01 \text{ kg/m}^2\text{.yr}$ for secondary forest for biotic production, occupation, vs. $-3.77\text{E-}02 \text{ kg/m}^2\text{.yr}$ from Bos et al. 2016).

Overall, larger discrepancies are obtained for land-transformation-related indicators. This may be explained by the differentiation of the weighting proxies between occupation and transformation CFs in the proposed upscaling technique (taking the respective flows in the developed inventory), as opposed to Bos et al. (2016), who used the same weights for upscaling the CFs for occupation and transformation flows. It is noteworthy that, unlike national occupation flows, which generally do not change considerably from one year to the next, the national transformation flows can vary significantly between years, with possible sign inversions due to different land use. This makes the resulting average CFs for transformation flows very sensitive to the year considered.

In practice, the use of different upscaling techniques can significantly influence the results of land use impact assessment. As examples, the global normalisation references calculated using the set of generic CFs documented by Bos et al. (2016) thus differed from those provided in Table 2 by factors ranging between 0.24 (erosion resistance) and 1.25 (mechanical filtration) for the five occupation-related impact categories and between 0.74 (mechanical filtration) and 26.2 (erosion potential) for the five transformation-related impact categories. These findings, therefore, demonstrate the need to harmonise the upscaling techniques for the land use impact category to provide a consistent set of generic CFs. This need can also be extended to other impact categories commonly assessed in life cycle assessment (Verones et al. 2017; Muttel et al. 2018).

3.4. Uncertainties, limitations and improvements potentials of the current inventory

Overall, four major sources of uncertainties were identified in the inventory – two of them limited to specific countries or territories (i.e. 1 and 2) while the other two are applicable to all countries (to different extent, depending on land class distribution in the country):

1. For small islands/territories, underestimations can be observed, i.e. typically with factors of 2-3 (see also Section 3.2.3 and Table S3). This is due to a lack of data for many land classes in those territories. Except at the local scale of these concerned countries, this uncertainty source is not deemed to have any influence on the results (e.g. continental, regional, global).
2. For countries having large territories in the polar regions, there are overestimations due to the projection used in the post-processing in QGIS software (WGS84), which enlarges the areas at the poles when the count is performed (i.e. counting of pixels of ca. 1 km²); this is particularly visible for a few countries like Greenland, Iceland, or Norway (see Table S3).
3. For all countries, there are potential overlaps, e.g. if within a same 1-km² cell, several land types co-exist. The applied counting technique (same as in previous inventory) consists in taking the most dominant land classes, with the assumption that it may be overestimated for some and underestimated in others. Discrepancies may occur as such. Likewise, “spatial overlaps” may also occur in the context of agroforestry, where tree and crop plantations co-exist on the same land. Differentiation between forestry and agricultural flows thus become difficult. Finally, “temporal overlaps” might also arise if the same land area serves different purposes in the course of the considered year, and thus end up counted several times under different land use classes. This is particularly relevant for arable land, for which part of the land can accommodate temporary crops in a season and fallow land in another.
4. Double counting may also occur between some of the land use data retrieved from FAOSTAT (2016) and the land cover data from FAO GLC SHARE (FAO 2014) –see also above. This may apply to some of the pasture/meadows, which could be regarded as herbaceous or sparse vegetation in the land cover maps, hence resulting in double counting and overestimations in the final inventory results (see Section 3.2.3). Such discrepancies, which are inherent to the combined use of land cover and land use data, are difficult to estimate and are much country-dependent.

Using these uncertainties and limitations, improvement potentials can then be defined. Further works should thus aim to harmonise the use of different data sources relying on both land cover mapping and land use inventories. Each data source has different nomenclatures, definitions and purposes; this currently results in significant inconsistencies when combined (e.g. see Prestele et al. 2017). In the context of life cycle impact assessment, land use data are required to better capture the extent of human activities on the land. However, land use data are difficult to find for all countries and land cover data from satellite observations remain useful in addition to exerting a harmonised inventorying methodology across all countries (Hansen et al. 2013). Such land cover map nonetheless renders differentiation between land classes difficult, e.g. no differentiation between types of forest or arable land (needed to match the level of disaggregation in the CFs). Hence, they cannot be used as stand-alone data sources in LCIA.

In addition, future research should investigate precise flow tracking to increase the consistency and level of details in the transformation flow inventories, for example differentiating the seven land use classes of shrub land, bare soil, herbaceous vegetation (aquatic or regularly flooded), sparse vegetation, mangroves, water bodies and snow/glaciers. Net data can typically be found or derived, like in this study, but data of where each transformation flow originates from and goes to, still remain unknown. This requires knowledge of what each grid cell is transformed from and to; such data could be gathered and processed from harmonised and sufficiently differentiated land use mapping over two consecutive years. It would allow for a comprehensive assessment of land transformation-related impacts, e.g. when assessing biodiversity impacts.

Finally, efforts should also be directed to expand the general coverage of land use flows in the inventory. Recently-developed LCIA methods have defined characterisation factors differentiated between crop types (e.g. Liu et al. 2018; Morais et al. 2018). Likewise, several existing methods differentiate the type of land management for agricultural and forestry land flows, for example between irrigated and non-irrigated land or between different tillage types (e.g. Mila i Canals et al. 2007; Brandao and Mila i Canals, 2013). To follow the pace of these LCIA method developments,

inventories of matching granularity are required. Readily available data for a top-down approach, as adopted in the current work, currently do not exist. This therefore calls for bottom-up modelling approaches, in which national and/or regional data on land use management practices should be collected and modelled to obtain more differentiated inventories in a global perspective.

4. Conclusions and recommendations

This study constitutes a first attempt at providing a globally-differentiated inventory of land occupation and transformation flows. A total of 40 land use flows representative for the year 2010 were thus inventoried for 225 countries in the world. The comparison of the developed inventory against previous land use inventories addressing specific regions or land use flows showed a relatively good agreement, hence attesting of its relative coherence. To the authors' knowledge, the built inventory is, therefore, the most comprehensive and up-to-date inventory, which has been published until now and is thus recommended for use in studies related to life cycle impact assessment.

To illustrate its use, global normalisation references for two LCIA methods for assessing land use impacts were calculated. Beyond the numerical values, this calculation highlighted the needs to refine the inventory, where future works should prioritise harmonising the combined use of land cover and land use data as well as expanding the inventory to further differentiate land classes (e.g. land transformation flows, crop types division) and land management types (e.g. irrigated/non-irrigated land). Furthermore, each country was found to be associated with specific patterns of land use impact intensities (when normalised by the country-specific total areas) across the range of existing land use indicators. This calls for a country-specific prioritisation when addressing land use impacts through policy-making, i.e. targeting the land use flows causing the largest environmental impacts at country level (identified from contribution analyses). To do so, further research is however needed to provide consistent approaches for ranking or aggregating land use impact

indicators. Likewise, the additional utilisation of the inventory for upscaling spatially-differentiated characterisation factors into generic CFs demonstrated the research need to consolidate the upscaling techniques and ultimately ensure that consistent and generic CFs are available to the LCA practitioners when the location of the land use is not precisely known.

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6. Electronic Supplementary Material

Contains an Excel file with two tables: (1) national inventories of land occupation and transformation flows (40 flows, incl. totals) for 225 countries in the world in 2010; (2) recalculated generic CF using the upscaling techniques based on the land use flows mapped in the developed inventory, (3) list of countries and territories with differences of more than 50% in the total occupation area.

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Tables

Table 1. Land use flows mapped in the inventory associated with data sources and the considered spatiotemporal coverage.

Major land use groups	Sub-groups	Spatiotemporal coverage	Data sources
Urban	1. Urban areas	World, country-specific; 2005, 2010	Hoskins et al. (2016); NASA (2016)
Agriculture	2. Agricultural areas 2.1. Arable land 2.1.1. Fallow land (temporary) 2.1.2. Temporary crops 2.1.3. Temporary meadows and pastures 2.2. Permanent crops 2.3. Permanent meadows and pastures	World, country-specific; 2005-2011	FAOSTAT (2016)
Forestry	3. Forest 3.1. Primary forest 3.2. Other naturally regenerated forest 3.3. Planted forest	World, country-specific; 2005-2011	FAOSTAT (2016)
	3.3.1. Extensive forest (planted forest) 3.3.2. Intensive forest (planted forest)	World, country-specific; 2005-2011	FAO (2010)
Other land use classes	4. Shrub land 5. Herbaceous vegetation (aquatic or regularly flooded) 6. Sparse vegetation 7. Bare soil 8. Mangroves 9. Snow and glaciers 10. Water bodies	World, country-specific; 1990-2012 ^a	FAO (2014)

^a combination of years 1990-2012 depending on the country (Latham et al., 2014).

Table 2. Global normalisation references for 2010 for the land use indicators defined in the LANCA method and the LCIA method by Milà i Canals (2007).

Indicator name	Occupation		Transformation	
	World	Unit	World	Unit
ILCD - Soil quality (Soil Organic Matter) ^a	- ^a	- ^a	1.28E+05 ^a	kg-C/person/yr
LANCA – Erosion Resistance	2.40E+04	kg/person/yr	3.68E+02	kg/yr/person/yr
LANCA – Mechanical Filtration	2.70E+05	m ³ /person/yr	1.47E+03	m ³ /yr/person/yr
LANCA – Physicochemical Filtration	8.58E+04	mol.yr/person/yr	9.15E+02	mol/person/yr
LANCA – Groundwater Regeneration	3.70E+02	m ³ /person/yr	7.19E-01	m ³ /yr/person/yr
LANCA – Biotic Production	2.81E+03	kg/person/yr	5.81E+00	kg/yr/person/yr

^a The method by Milà i Canals et al. (2007) defines one single indicator when characterising both occupation and transformation flows, hence only one normalisation reference is provided for the entire impact category (all land use flows considered).

Figure captions

Figure 1. Overall methodology for inventory building and calculation of normalisation references.

Figure 2. Comparison of occupation data for urban areas between the current study and other data sources at **(a)** global and **(b)** European levels. Data sources: Atlas of Urban Expansion (Lincoln Institute of Land Policy, 2016) and MODIS (Natural Earth, 2016) for global coverage; EU-27 inventory (Sala et al., 2014) and CORINE Land Cover (EEA, 2014) for Europe. Each data source has a different country coverage (countries covered in current study are plotted here). Years of the inventories indicated in the figure legends are the years specific to the inventory data, and not the year of publication. Linear correlation results (r-squared values) are indicated in the figure legends for each compared dataset. Note the logarithmic scale on the axes.

Figure 3. Comparison of occupation data between the current study and other data sources for **(a)** total forest area at global scale and **(b)** at EU level, **(c)** total pasture area at global scale, **(d)** total estimated cropland (i.e. arable land, plus permanent crops, minus temporary pasture/meadows) at global scale, and **(e)** total arable land at EU level. Different data sources are used for different land classes: Hansen et al. (2013) for forestry areas, Hoskins et al. (2016) for pasture and cropland areas, and Sala et al. (2014) for EU forestry and arable land (original source: UNFCCC). Linear correlation results (r-squared values) are indicated in the plot for each compared dataset. Note the logarithmic scale on the axes.

Figure 4. Comparison of occupation data between the current study and other data sources for total country area at global scale. Data sources: FAOSTAT (2018) and CIA (2017) for total country areas. Linear correlation results (r-squared values) are indicated in the figure legend for each compared dataset.

Figure 5. National land use impacts normalised by total country area for **(a)** groundwater regeneration (occupation, LANCA), **(b)** groundwater regeneration (transformation, LANCA), **(c)** erosion resistance (occupation, LANCA), **(d)** biotic production (occupation, LANCA), **(e)** physicochemical filtration (occupation, LANCA), and **(f)** soil organic matter (both occupation and transformation, Milà i Canals et al. 2007).

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