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# An improved life cycle impact assessment principle for assessing the impact of land use on ecosystem services

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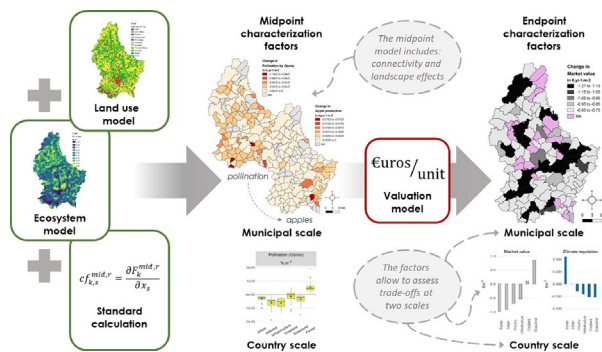
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## HIGHLIGHTS

- A new land use (LU) impact characterization approach on ecosystem services is built
- We use a spatial and dynamic integrated model of LU and ecosystem services (ES)
- New characterization factors assess trade-offs and synergies among ES
- The “landscape effect” of land cover changes is taken into account
- Factors are regionalized at country and municipality scales for Luxembourg

## GRAPHICAL ABSTRACT



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## ABSTRACT

In order to consider the effects of land use, and the land cover changes it causes, on ecosystem services in life cycle assessment (LCA), a new methodology is proposed and applied to calculate midpoint and endpoint characterization factors. To do this, a cause-effect chain was established in line with conceptual models of ecosystem services to describe the impacts of land use and related land cover changes. A high-resolution, spatially explicit and temporally dynamic modeling framework that integrates land use and ecosystem services models was developed and used as an impact characterization model to simulate that cause-effect chain. Characterization factors (CFs) were calculated and regionalized at the scales of Luxembourg and its municipalities, taken as a case to show the advantages of the modeling approach. More specifically, the calculated CFs enable the impact assessment of six land cover types on six ecosystem functions and two final ecosystem services. A mapping and comparison exercise of these CFs allowed us to identify spatial trade-offs and synergies between ecosystem services due to possible land cover changes. Ultimately, the proposed methodology can offer a solution to overcome a number of methodological limitations that still exist in the characterization of impacts on ecosystem services in LCA, implying a rethinking of the modeling of land use in life cycle inventory.

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## 1. Introduction

The use of land for human activities induces land cover changes that affect ecosystems in many ways. Their degradation induces trade-offs among ecosystem services, i.e. the benefits that we retrieve from their functionality, such as resources (e.g., crops, timber, water), the regulation of climate, or culture (Costanza et al., 2017). Given the immense value of ecosystem services, some of them being both essential and irreplaceable (Costanza et al., 2014), and the current decline of the biodiversity that supports their provision (Butchart et al., 2010), the future development of societies should take them into account and attempt to preserve them as much as possible. Several authors promote the impact assessment of land use on ecosystem services in life cycle assessment (LCA), as a mean to include their valuation in decision practices (Chaudhary et al., 2017; Knudsen et al., 2017; Koellner et al., 2013b; Othoniel et al., 2016; Pavan and Ometto, 2018; Taelman et al., 2016). Life-cycle uses of land encompass, for instance, the use of arable land for producing crops for biofuels (Maia de Souza et al., 2018; Vázquez-Rowe et al., 2014).

LCA typically addresses the impact of a stressor on an indicator in a linearized way, using a “characterization factor” (CF) that expresses the marginal change in the indicator due to a marginal change in the stressor. As of today, several research streams focus on calculating CFs for the assessment of land use impacts on ecosystem services, which is the scope of our study. The main stream surrounds the framework selected as a reference by the UNEP/SETAC for harmonizing global land use impact assessment on biodiversity and ecosystem services in LCA (Koellner et al., 2013b; Lindeijer, 2000; Milà et al., 2007). This framework established the methods that are commonly used for the calculation of CFs, our focus, and the modeling of land use in life cycle inventory (LCI). Numerous studies have calculated CFs following this framework (e.g., Brandão and I Canals, 2013; De Baan et al., 2013; Müller-Wenk and Brandão, 2010; Saad et al., 2013; Taelman et al., 2016; van Zelm et al., 2018). However, these CFs have not yet reached consensus, and assessing land use impacts on ecosystem services is still not a standard practice in LCA, as observed by several authors (Jeswani et al., 2018; Maia de Souza et al., 2018; Othoniel et al., 2016; Pavan and Ometto, 2018; Teixeira, 2014). In summary, three main challenges remain for the calculation of CFs. First, the dynamics and connectivity of ecosystems and humans among socio-ecological systems should be simulated when calculating CFs. Second, CFs should be regionalized using a multiscale approach, in order to a) identify the benefits that are provided by ecosystems at a local scale, and b) obtain CFs that are compatible with the regionalization of LCI (typically at the country scale). Third, the impact indicators that are calculated thanks to such CFs should reveal, in terms of ecosystem services, the costs and benefits to society induced by land use.

In this line, two alternatives to the UNEP/SETAC framework have been proposed to characterize the impact of land use on ecosystem services. The first alternative, illustrated by Arbault et al. (2014) and Liu et al. (2018) and to some extent by others (Bare, 2011; Othoniel et al., 2016), relies on the use of system dynamics modeling to simulate the dynamics of the earth system, including functioning ecosystems, land use and other human activities. Although this method seems promising, the models used in these studies were not spatially resolved. Hence, they could not simulate the effect of changing land cover patterns on the functionality of ecosystems (which some authors call the “landscape effect” of land cover changes), although it is relevant for assessing impacts ecosystem services (Baude et al., 2019; Bennett and Isaacs, 2014; Pufal et al., 2017; Ricketts et al., 2008). Clermont et al. (2015), for instance, observed this effect in the case of urbanization impacting honey bees in Luxembourg. The second alternative, proposed by Chaplin-Kramer et al. (2017), uses spatially-resolved models of land cover changes and functional ecosystems to assess the impacts of LCI results on ecosystem services. They modeled the spatial effects of land cover changes and showed the potential advantages of using a high-

resolution land cover model to calculate CFs. However, they did not calculate CFs, but instead used their model as a “plugin” to combine with LCA software. Out of our scope, we mention for the sake of completeness that several authors have argued for the accounting of changes in ecosystem services in LCI (when environmental stressors are accounted for) instead of assessing impacts on ecosystem services with CFs (Bakshi et al., 2015; Blanco et al., 2018; Liu et al., 2018; Zhang et al., 2010).

In this paper, we design a new approach for calculating CFs that can be used in LCA to assess the impact of land use on ecosystem service. A first novelty of our approach lies in the model that we used for characterizing midpoint impacts. More specifically, we developed an integrated system dynamics model of land cover and ecosystem services. We did so following the Multiscale Integrated Model of Ecosystem Services (MIMES) framework (Boumans et al., 2015), an acknowledged reference for the modeling of socio-ecological systems (Balvanera et al., 2017; Turner et al., 2015). A second novelty lies in how this model is used, i.e., the formulas followed for applying it and summarizing its results. In particular, we calculated CFs that quantify the trade-offs induced by land cover changes among ecosystem services when one additional unit of land is used. An underlying assumption of our approach is that we characterize the impacts of marginal demands for land covers, which are going to be met by land cover changes, given the current scarcity of land (Lambin and Meyfroidt, 2011). This assumption, also made by Chaplin-Kramer et al. (2017) is not compatible with the current modeling of land use in LCI. According to this latter, land use should be modeled with two distinct stressors, for which dedicated CFs are calculated, while we assumed the modeling of land use with a single stressor.

We applied our approach to the Grand Duchy of Luxembourg and computed regionalized CFs at both the national scale, for the whole country, and the local scale, for its municipalities. We thus could test if using an integrated model allows calculating CFs that are more representative of the mechanisms underlying the impact of land use and land cover changes on ecosystem services, including landscape effects, interactions among ecosystem functions, and several constraints of land use. As we will see, our approach indeed allows to take these aspects into account in the calculation of CFs. However, it requires the implementation of complex models. Therefore, the validity and uncertainty of the CFs should be thoroughly assessed for their integration in decision-making. In our case, the validity of the models that we used implies that further modeling efforts are still needed in order to build the credibility of the CFs and make them fully operational.

## 2. Materials and methods

### 2.1. Study area and data

To test our approach, we calculated CFs for the Grand Duchy of Luxembourg (country scale) and its 116 municipalities (local scale). In Luxembourg, land is a very scarce resource. Almost all of its 2500 km<sup>2</sup> of land are intensively used or not designated for conversion. In 2016, 50% of land was covered by agriculture, 35% by forest areas, 14.5% by built areas (urban areas, industries, infrastructures) and 0.5% by water areas (STATEC, 2019). This is a relevant issue since the Luxembourgish authorities have stated several political objectives that will imply the further use of land. First, as the country wishes to become more attractive to foreign workers, the population is expected to double by 2060, from 0.5 to 1 million inhabitants (Hennani, 2017). As a result, urban areas and transportation infrastructures should increase. Furthermore, as a mean to meet European regulations on the emission of greenhouse gasses, the government aims to increase the domestic production of renewable energy, according to their recently published strategic plan “Luxembourg 2030” (Stoldt Associés, 2018). Depending on the selected option, this may imply the expansion of agricultural areas and infrastructures. Finally, under

the MAES European initiative<sup>1</sup> (European Environment Agency, 2015) and other international texts (Stoldt Associés, 2018), Luxembourg should establish new environmental protection measures in order to preserve its ecosystems and the services that they provide. Ultimately, this should create a saturated situation, where any decision to further use land must be thoroughly evaluated prior to its concretization. In this context, the CFs provided in this study could be useful means of environmental impact assessment if the authorities use LCA to assess their development plans.

In addition to being an interesting case, the fact that Luxembourg is small eases the implementation of models. We thus could gather the data that are listed and depicted in Table 1 and Supplement A.4. Thanks to these, we could simulate land use and ecosystem services at the acceptable resolution of 500 m. We could also consider several constraints of land use changes, such as protected areas (e.g., national reserves, Natura 2000) and the slope of land (e.g., urban areas tend to expand on flat land). Finally, we could use observed data for the evaluation of some ecosystem services, which is a positive aspect (Egoh et al., 2012).

## 2.2. Characterization factors: rationale, calculation and use

A CF is a local derivative of a stressor-impact (or dose-response) model, that can be used in a first-order Taylor approximation of the outcome of the model by means of a linear estimate (Frischknecht and Jolliet, 2016; Heijungs and Suh, 2002; Pennington et al., 2004). The original non-linear model is called a “characterization model”. As a general model structure, we can conceive the impact as an indicator with the initial value  $m$  as a result of different amounts of  $S$  types of stressors (vector  $\mathbf{x}$  of lengths  $S$ ) as follows:

$$m = F(\mathbf{x}) \quad (1)$$

where  $F(\cdot)$  is the characterization model that represents the cause-effect function linking the amounts of the stressors ( $\mathbf{x}$ ) to the impact ( $m$ ).

In the context of a marginal change in one or more stressors, we can approximate the change in impact (impact indicator) as:

$$\Delta m \approx \sum_{s=1}^S \left. \frac{\partial F}{\partial x_s} \right|_{\mathbf{x}=\mathbf{x}_{current}} \times \Delta x_s \quad (2)$$

where  $\Delta m$  is the impact indicator;  $\Delta x_s$  represents a change in the  $s$ -th stressor; and  $\mathbf{x} = \mathbf{x}_{current}$  indicates that the partial derivative is calculated at the current level of  $\mathbf{x}$ .

The partial derivative  $\left. \frac{\partial F}{\partial x_s} \right|_{\mathbf{x}=\mathbf{x}_{current}}$  is what we refer to as the characterization factor (Heijungs and Suh, 2002):

$$cf_s = \left. \frac{\partial F}{\partial x_s} \right|_{\mathbf{x}=\mathbf{x}_{current}} \quad (3)$$

where  $cf_s$  is the CF that assesses the impact of the  $s$ -th stressor.

Therefore, CFs are literally derived from a characterization model through differentiation. In Eqs. (2) and (3), the derivative is taken at  $\mathbf{x} = \mathbf{x}_{current}$ , the current levels of the stressors. As a result, CFs address change-oriented issues that are relevant for “now”, which can mean today, this year or this decade, depending on how the current level of the stressor was measured (which time period) and how fast the level of the stressor changes in reality.

Once CFs have been calculated for the  $S$  stressors, they serve to translate changes in stressors into an impact indicator as follows:

$$\Delta m \approx \sum_{s=1}^S cf_s \times \Delta x_s \quad (4)$$

When  $F(\cdot)$  is non-linear, which is the case for the impact of land use on ecosystem services (Koch et al., 2009), Eq. (4) only makes sense under the assumption of marginal  $\Delta x_s$ . Therefore, the use of CFs is not necessarily valid for non-marginal changes.

In LCA, the impact of a stressor is assessed, from a conceptual point of view, along multiple cause-effect chain models (which are assumed independent, although they may interact) that each depict the relations between a midpoint indicator that depends on the stressor, and an endpoint indicator that depends on the midpoint indicator. As an example of the cause-effect chain, the impact of CO<sub>2</sub> emissions (stressor) is assessed on radiative forcing (midpoint) and human health (endpoint that depends on global warming). Midpoint indicators should represent specific ecological issues, like radiative forcing, while endpoint indicators should represent primary dimensions of well-being, like human health (Hauschild et al., 2018; Hauschild and Huijbregts, 2015; Jolliet et al., 2014). In practice, two types of CFs, namely midpoint CFs and endpoint CFs, are used, following Eq. (4), to translate changes in stressors into changes in midpoint and endpoint indicators, respectively. Midpoint CFs are calculated following Eq. (3) and using a midpoint characterization model, which we will denote as  $F^{mid}(\cdot)$ . Endpoint CFs are calculated in two steps: 1) mid-to-endpoint CFs are calculated following Eq. (3) and using an endpoint characterization model, which we will denote as  $F^{mid-to-end}(\cdot)$ ; and 2) endpoint CFs are calculated from the aggregation of midpoint and mid-to-endpoint CFs. In the Supplement A.1 (Supplementary data), we adapt Eqs. (3)–(4) to the cases of midpoint and endpoint CFs and formalize the calculation of endpoint CFs.

In the present study, we calculated midpoint and endpoint CFs that assess the impacts of demands for the  $S = 6$  land covers on the  $K = 6$  ecosystem functions (midpoint indicators) and  $I = 2$  ecosystem services (endpoint indicators) that are listed in Table 2. The four key terms “land use”, “land cover”, “ecosystem function” and “ecosystem service” are defined in the Supplement A.2. We selected the items in Table 2 mainly because of the larger availability of data and model compared to other possible ecosystem functions and services. In the discussion and the Supplement A.3, we introduce some classifications to expand the calculation of CFs (increase  $S$ ,  $K$  and  $I$ ).

## 2.3. Cause-effect chain, characterization models and choices of indicators

To calculate our CFs, we developed a midpoint characterization model and an endpoint characterization model following the three-step conceptual cause-effect chain illustrated in Fig. 1. In the first step, demands for land covers, our stressors, influence the state of land cover and induce land cover changes. In the second step, the state of land cover influences the functionality of ecosystems and therefore the production of ecosystem functions, which are our midpoint indicators, measured on an annual basis. We simulated both steps using an integrated model of land cover changes and ecosystems functioning as explained in the section “Midpoint characterization model”. Finally, in the third step, the values of ecosystem services (benefits that are actually provided by ecosystem functions) depend on the supply of ecosystem products, which is called the “flows of ecosystem services” (Serna-Chavez et al., 2014). The values of ecosystem services are our endpoint indicators, also measured per year. We valued the benefits of ecosystem services using monetary estimates in euros, as explained in the section “Endpoint characterization model”.

We selected our midpoint and endpoint indicators on the basis of the “cascade model” for ecosystem services (Potschin-Young et al., 2018), as recommended by Maia de Souza et al. (2018) and Pavan and

<sup>1</sup> Mapping and Assessment of Ecosystem Services: an initiative of the European Union that compels the participating countries to provide national maps and assessments of the states of land, biodiversity, ecosystems, and ecosystem services. These data, which we used in part in our study, are publicly available (European Environment Agency, 2015).

**Table 1**

Data that were used as input in the midpoint characterization model. All the spatial were sampled according to the same square grid with 500 m resolution (res).

Data	Description	Format	Submodel	Source
Land cover	Satellite observation of land's coverage (CORINE dataset) for the years 1990, 2000, 2006 and 2012	Raster (res: 100 m)	Land use	European Environment Agency (2019)
Roads	Network of roads, extracted from the Luxembourg's OBS data (observed biophysical coverage of the soil)	Shapefile	Land use	Administration du Cadastre et de la Topographie (2007)
Slope	Slope of the land, calculated from the elevation map of Luxembourg	Raster (res: 10 m)	Land use	Administration du Cadastre et de la Topographie (2007)
Protected areas	Land areas that are protected for the conservation of biodiversity (e.g., national reserves, Natura 2000)	Shapefile	Land use	Administration du Cadastre et de la Topographie (2007)
Foraging resources and nesting suitability	Expert-based indicators that describe the quality of land covers as habitats and as sources of food for pollinators	Numerical	Pollination	Zulian et al. (2013)
Types of pollinators	Physiological data (e.g., flying distance) for the two modeled species of pollinators: <i>Osmia</i> and <i>Bombus</i>	Numerical	Pollination	Greenleaf et al. (2007)
Apples' dependency on pollinators	Expert-based indicator that expresses by how much yields depend on pollination (90% in the case of apples)	Numerical	Pollination	Zulian et al. (2013)
Apple orchards	Areas of apple orchards, extracted from the Luxembourg's OBS data (observed biophysical coverage of the soil)	Shapefile	Pollination	Administration du Cadastre et de la Topographie (2007)
Soil type	Type of soil per cell of the model, according to the Soil Geographical Database of Eurasia (SGDBE), version 4 beta (8 classes are present in Luxembourg)	Shapefile	Carbon sequestration	The European Soil Database (2001)
Meteorological data	Daily measurements of solar radiation, rainfall and temperature for 1950 onwards (one station for the whole Luxembourg)	Numerical	Carbon sequestration	Findel airport SYNOP station (WMO station ID = 06590)
Physiological parameters	Physiological characteristics of C3 grasses and the main tree species in Luxembourg: <i>Picea abies</i> , <i>Fagus sylvatica</i> , <i>Pinus sylvestris</i> , <i>Quercus robur/petraea</i>	Numerical	Carbon sequestration	Pietsch et al. (2005)
MAES data	Results of statistical surveys for the production of crops and livestock, mapped over the whole Luxembourg	Raster (res: 500 m)	Crop and livestock production	Becerra-Jurado et al. (2015)

Ometto (2018). This model, which is designed in a cause-effect fashion like our own cause-effect chain, depicts ecosystem services depending on ecosystem functions, which arise from biological components (Potschin-Young et al., 2018). Land covers represent these latter in our case. Ecosystem functions are hence "in the middle of" land covers (source of the beneficial products) and ecosystem services (benefits to people). In addition, impacts on ecosystem functions such as pollination represent ecological issues, which midpoint impact indicators should do (Hauschild and Huijbregts, 2015). Likewise, impacts on ecosystem services such as food production represent direct impacts on people's

well-being, which endpoint impact indicators should do (Hauschild and Huijbregts, 2015). Notice that in our study, we assess both negative (costs) and positive (benefits) impacts.

In this paper, we calculated CFs that are regionalized at two scales, for Luxembourg as a whole and for each of its 116 municipalities. In this regard, note that we considered only the impacts that occur within Luxembourg or a given municipality, i.e., we omitted potential transboundary effects. Also, we summed over the modeled territory the production of ecosystem functions and the values of ecosystem services when calculating CFs (Fig. 1).

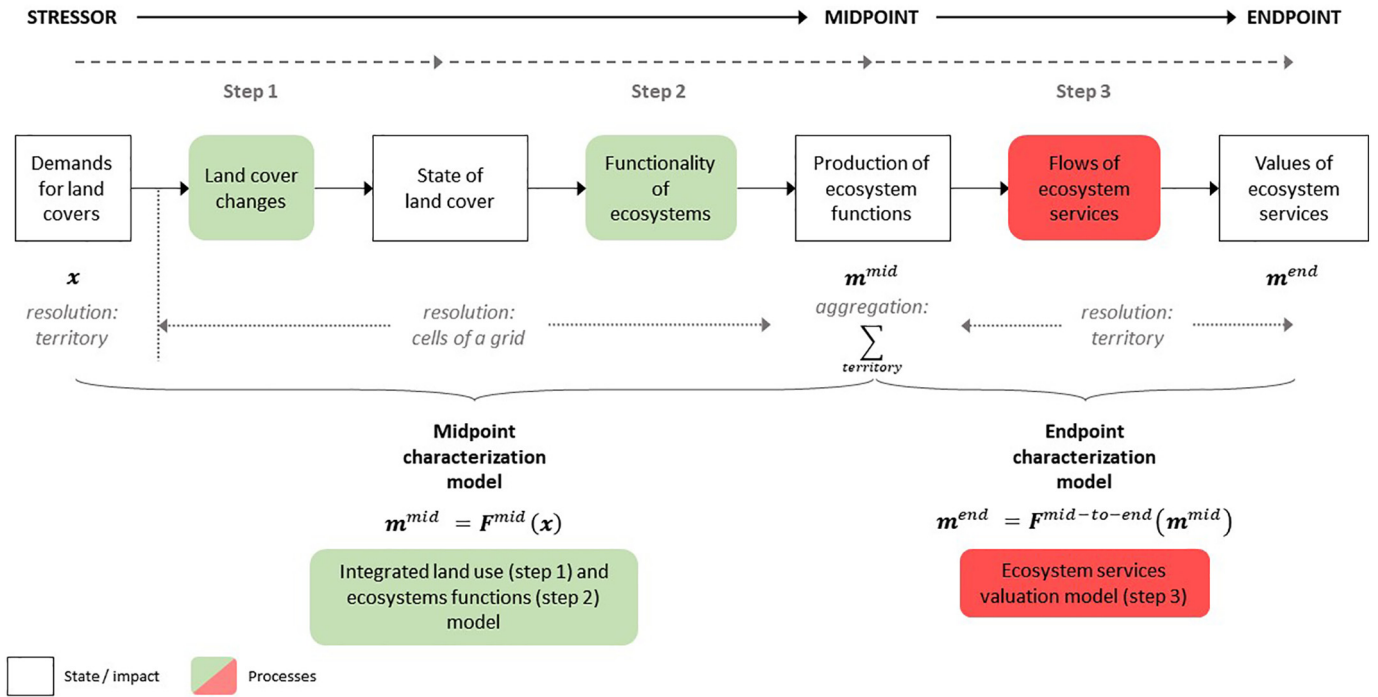
**Table 2**

typologies of modeled land covers (LC), ecosystem functions (EF) and ecosystem services (ES). A total of  $6 \times 6 = 36$  midpoint (LC x EF),  $6 \times 2 = 12$  mid-to-endpoint (EF x ES) and  $6 \times 2 = 12$  endpoint (LC x ES) sets of characterization factors are calculated accordingly.

Typology	Type	Description	Unit <sup>a</sup>
Land covers (stressors)	Urban	Built-up areas that are used for housing and other activities in cities (areas with a relatively high population density), such as transport.	m <sup>2</sup>
	Industry	Built-up areas that are used for manufacturing activities.	m <sup>2</sup>
	Infrastructure	Built-up areas that are used for transportation, such as roads, railways or airports.	m <sup>2</sup>
	Cropland	Areas that are used for agricultural purposes other than permanent crops (orchards and vineyards).	m <sup>2</sup>
	Grassland	Grassy and non-woody fields that are either used as meadow (~15% in Luxembourg) or pasture (~85%).	m <sup>2</sup>
	Forest	Woody areas. In Luxembourg, they are mostly preserved and used for recreational purposes.	m <sup>2</sup>
Ecosystem functions (midpoint indicators)	Pollination ( <i>Osmia</i> )	The activity of wild pollinators that forage for food and fertilize pollination-dependent crops on their way.	.
	Pollination ( <i>Bombus</i> )		yr <sup>-1b</sup>
	Apple production	Capacity of orchards to produce apples.	kg. yr <sup>-1</sup>
	Crop production	Capacity of agricultural areas to produce arable crops.	kg. yr <sup>-1</sup>
	Livestock production	Capacity of grassland (pastures) to support the husbandry of the livestock.	LSU. yr <sup>-1</sup>
	Carbon sequestration	Capacity of plants to fix and store carbon in soils and, to some extent, in their woody parts.	kgC. yr <sup>-1</sup>
Ecosystem services (endpoint indicators)	Total market value	Economic benefits due to the marketing of ecosystem products (food and material).	€. yr <sup>-1</sup>
	Climate regulation	Benefits (or avoided costs) to well-being (in terms of health, production of energy and food, protection against extreme events, etc.) that are due to the natural regulation of climate within liveable and expectable standards.	€. yr <sup>-1</sup>

<sup>a</sup> The units come from the models that were used in the midpoint characterization model. These models are described in the Supplement A.4.

<sup>b</sup> While pollination is modeled with a dimensionless indicator, its change is measured in % when calculating CFs, as detailed in the Supplement A.5.1.



**Fig. 1.** Three-step cause-effect chain from demands for land covers (stressor  $x$  that is accounted for in life cycle inventory) to the values of ecosystem services (endpoint indicator  $m^{end}$ ). The bottom part represents the characterization models ( $F^{mid}$  and  $F^{mid-to-end}$ ) that were implemented to simulate the midpoint and endpoint impacts in this paper. We consider the total production over a territory of ecosystem functions ( $m^{mid}$ ) and values of ecosystem services ( $m^{end}$ ) as midpoint and endpoint indicators, while we rely on the simulation of land cover and ecosystem functions at a higher resolution (cells of a grid). Rectangles denote states or impacts, rounded colored rectangles denote processes that connect such states, green represents the part between stressor and midpoint, red represents the part between midpoint and endpoint. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

#### 2.4. Midpoint characterization model

Our midpoint characterization model covers the first two steps of Fig. 1 and is an integrated model that couples:

- The model CLUE (Verburg and Overmars, 2007, 2009) for simulating land cover and its changes (step 1 in Fig. 1). This simulates over a spatial grid the growth and the conversion of land cover patches (cf. Supplement A.4.2).
- The model InVEST (Lonsdorf et al., 2009; Zulian et al., 2013) for pollination and apple production (part of step 2). This simulates the presence of pollinators over a grid, their activity in orchards and their contribution to apple yields (cf. Supplement A.4.3.1).
- The model BIOME-BGC (Thornton et al., 2005) for carbon sequestration (part of step 2). This simulates the growth of forests and their exchanges of materials with their environment on a daily basis using meteorological data (cf. Supplement A.4.3.2);
- The spatial statistical datasets generated within the Luxemburgish Mapping and Assessment of Ecosystems and their Services (MAES) initiative (Becerra-Jurado et al., 2015) for the production of crops and livestock (part of step 2; cf. Supplement A.4.3.3).

We implemented this integrated model in a system dynamics environment following the MIMES modeling framework. We did so using the Simile (Muetzelfeldt and Massheder, 2003) and R (R Core Team, 2018) software. More details about MIMES, the constituent models and data can be found in the Supplement A.4. The Simile model file is available as Supplementary material (cf. Supplement A.4.4).

In this study, we used the year 2012 as the reference situation (so for  $x_{current}$ ) because all the data that we needed were unavailable for later years. CFs shall be updated over time when new data are available.

For calculating municipality scale midpoint CFs, we applied our midpoint characterization model  $F^{mid}(\cdot)$  to each of the  $R = 116$  municipalities of Luxembourg. When parametrized for a municipality  $r$ , this

model allows to simultaneously simulate land cover changes and the total production of the  $K$  ecosystem functions in response to driving demands for land covers:

$$m^{mid,r} = F^{mid,r}(x) \tag{5}$$

where  $F^{mid,r}(\cdot)$  is the midpoint characterization model parametrized for  $r$ ;  $m^{mid,r}$  (vector of length  $K$ ) represents the total production of the ecosystem functions in  $r$ ; and  $x$  (vector of length  $S$ ) represents the demands for the land covers.

Because we could not disentangle the analytical form of  $F^{mid}(\cdot)$  (too complex), we could not analytically calculate its partial derivative as Eq. (3) would suggest. Therefore, we numerically approximated its derivatives, calculating the municipality-scale midpoint CFs as follows:

$$cf_{k,s}^{mid,r} = \left. \frac{\partial F_k^{mid,r}}{\partial x_s} \right|_{x=x_{current}} \approx \lim_{\Delta x_s \rightarrow 0} \frac{\Delta m_k^{mid,r}}{\Delta x_s} \tag{6}$$

where  $cf_{k,s}^{mid,r}$  is the midpoint CF that assesses the impact of expanding land cover  $s$  on the ecosystem function  $k$  in the municipality  $r$ ;  $x_s$  is the demand for  $s$ ; and  $\Delta m_k^{mid,r}$  is the change in the total production of  $k$  in  $r$  that is due to the change  $\Delta x_s$  in the demand for  $s$ . We took  $\Delta x_s$  as small as possible, which in our case corresponds to the 1 ha resolution of our model.

Eq. (6) was done for all land cover types  $s$ , ecosystem functions  $k$  and municipalities  $r$  of Luxembourg. We thus calculated  $6 \times 6 \times 116 = 4176$  of such midpoint CFs. Be aware that for calculating all the CFs for a given land cover  $s$  ( $cf_{k,s}^{mid,r}$  for all  $k$ ), we performed a unique simulation of the model in  $r$ . Thanks to this, we could encompass potential interactions between ecosystem functions in the calculation of midpoint CFs (in our case, apple production depends on pollination). A more detailed description of the model function and of the approximation of the midpoint CFs can be found in the Supplement A.5. Also, we converted the

values of the CFs into values per m<sup>2</sup>, the standard unit for land areas in LCI.

Finally, we calculated country-scale midpoint CFs as the arithmetic mean of the municipal CFs, excluding a few municipalities for which we did not compute CFs for some land cover types since we could not calibrate the model in the absence of past observed changes in these municipalities (see the Supplement 4.2 for details):

$$c_{f_{k,s}}^{mid,lux} = \frac{1}{R} \sum_{r=1}^R c_{f_{k,s}}^{mid,r} \quad (7)$$

where  $c_{f_{k,s}}^{mid,lux}$  is the midpoint CF that assesses the impact of expanding land cover  $s$  on the ecosystem function  $k$  for the whole country of Luxembourg.

Implicit in this is an equal weighting of municipalities; see the Discussion for some remarks on this.

Fig. S-6 in the Supplement A.5 illustrates the workflow for calculating municipality and country-scale CFs, taking as an example the impacts of urban expansion on pollination by *Osmia* and apple production.

### 2.5. Endpoint characterization model

This section depicts the endpoint characterization model that we implemented to calculate mid-to-endpoint CFs according to Eq. (3), which we then used to calculate endpoint CFs at both municipality and country scales (cf. the Supplement A.1).

For modeling the relation between the production of ecosystem functions and the benefits that they provide (Step 2 in Fig. 1), we valued the provision of ecosystem services in monetary terms (euros), like Cao et al. (2015).

As first endpoint indicator, we considered the total market value of the products of ecosystem functions, which is equal to:

$$m_1^{end} = F_1^{mid-to-end}(\mathbf{m}^{mid}) = \sum_{k=1}^6 p_k \times m_k^{mid} \quad (8)$$

where  $F^{mid-to-end}(\cdot)$  is the endpoint characterization model;  $m_1^{end}$  is the total market value of ecosystem products;  $m_k^{mid}$  the production of ecosystem function  $k$  (the  $k$ -th element of  $\mathbf{m}^{mid} = \sum_r \mathbf{m}^{mid,r}$ ) and  $p_k$  the market price of its product (see Table 3). We set the market price of the product of pollination by *Osmia* and *Bombus* to 0, since we assumed that pollination is a fully intermediary function, the benefits of which are already encompassed in the benefits of apple production. The market price of the product of carbon sequestration is also equal to 0.

**Table 3**

Mid-to-endpoint characterization factors (CFs) applied to calculate endpoint CFs for the impact assessment of land use on ecosystem services in Luxembourg.

Endpoint indicator	Variable	Midpoint indicator	Value	Unit	Data source
Total market value	$\mathbf{p}$	Pollination ( <i>Osmia</i> )	0	€ <sup>a</sup>	STATEC (2019) & (data refer to year 2012)
		Pollination ( <i>Bombus</i> )	0	€ <sup>a</sup>	
		Apple production	2.5	€/kg	MAES (Becerra-Jurado et al., 2015)
		Crop production	2	€/kg	
		Livestock production	2500	€/LSU	
		Carbon sequestration	0	€/kgC	
Climate regulation	$\mathbf{q}$	Pollination ( <i>Osmia</i> ) <sup>b</sup>	0	€	Carbon price estimate (World Bank, 2019) evaluated in €/kgC
		Pollination ( <i>Bombus</i> ) <sup>b</sup>	0	€	
		Apple production	0	€/kg	
		Crop production	0	€/kg	
		Livestock production <sup>b</sup>	0	€/LSU	
		Carbon sequestration	0.02	€/kgC	

<sup>a</sup> The unit for pollination is in € because it is modeled with a dimensionless indicator.

<sup>b</sup> Since we consider impacts on climate regulation and not climate change effects, we do not consider the potential emissions of C by animals, although they can be considered as an ecosystem "disservice" (Schaubroeck, 2017; Von Döhren and Haase, 2015).

Therefore, we calculated a mid-to-endpoint CF for each product  $k$  as follows:

$$c_{f_{1,k}}^{mid-to-end} = \frac{\partial F_1^{mid-to-end}}{\partial m_k^{mid}} = p_k \quad (9)$$

where  $c_{f_{1,k}}^{mid-to-end}$  is the endpoint CF that assesses the impact of a change in the production of  $k$  on  $m_1^{end}$ .

Moreover, we followed the same method to calculate mid-to-endpoint CFs for impacts on climate regulation (second endpoint indicator). In this case, only carbon sequestration provides climate regulation benefits (avoided social costs), which we approximated with data on carbon prices,  $\mathbf{q}$  (see Table 3). So

$$c_{f_{2,k}}^{mid-to-end} = \frac{\partial F_2^{mid-to-end}}{\partial m_k^{mid}} = q_k \quad (10)$$

Table 3 lists all the monetary values that we used, with their sources. We used national estimates of ecosystem services values due to a lack of data at the municipality scale.

### 2.6. Validation of the models

As a part of the midpoint characterization model's development (cf. Supplement A.4.2.4), we validated its constituent models – i.e., we assessed their capacity to predict future changes (Oreskes et al., 1994). The primary focus is on the validation of the land use model, as depicted below. Concerning the BIOME-BGC model, we used the parameters that were validated by Pietsch et al. (2005). In addition, we visually compared its output (gross primary productivity, for which we had observations) with Eddy covariance measures (FLUXNET, 2018). The procedure for selecting the observed data is detailed in the Supplement A.4.3.2. For InVEST pollination, we compared the total yield over Luxembourg that it simulates with data reported by the Luxemburgish office of statistics (STATEC, 2019). We lacked spatial data on orchards yields in order to test its spatial output. Finally, the MAES data do not require a validation since they are based on observations from statistical survey.

In order to validate the land use model, we mostly relied on the relative operating characteristic (ROC) method, promoted by Pontius and Parmentier (2014). The ROC is a method that allows to test a Boolean classifier based on a [0,1] index. In our case, the Boolean classifier is the land use model that decides if a given land cover expands in a cell or not (Boolean), and the [0,1] index is the suitability of cells for this land cover (an intermediary variable of our model, as explained in the Supplement A.4.2.3). This method yields an ROC curve, like the one plotted in the bottom right of Fig. 2. The main statistic that is used to communicate the results of the model's diagnosis is the area under the

curve, which is between [0,1] with 1 that indicates a perfect classifier and 0.5 a theoretical random model. However, this measure suffers certain limitations, and therefore should be complemented others to thoroughly evaluate the accuracy of the model. For an exhaustive and formal description of the ROC, its limitations and its qualities, consult Pontius and Parmentier (2014). In addition to the ROC, we also compared and mapped (for visual validation) the count and spatial correspondence of changed spatial units between the simulated and observed data. Fig. 2 exemplifies the validation of the model's capacity to predict the expansion of infrastructure in a single municipality. The model was calibrated over the period 1990–2000 and validated over the period 2000–2012.

Concerning the endpoint characterization model, since it is the monetarized version of the midpoint model, its validity depends on the validity of this latter. Therefore, it does not require an additional validation. The only additional input data that were used in the endpoint characterization model are unit prices, which come from statistical surveys (cf. Table 3).

### 2.7. Statistical analysis

To test if the impacts on two ecosystem functions or services are correlated, we calculated the Spearman's rank correlation coefficient, denoted as  $\rho$ . This nonparametric measure assesses if the two variables are monotonically related – i.e., if their ordered observations have the same rank. The value of  $\rho$  is between  $-1$  and  $+1$ . A positive value indicates a positive correlation, which in our case means that there is a synergy between the impacts on the two ecosystem functions/services. A negative value means, in our case, that there is a trade-off (negative correlation) between the impacts. Typically, we consider that the test is significant when its  $p$ -value is  $p \leq 0.05$ .

## 3. Results

### 3.1. Midpoint characterization factors for Luxembourg

All the calculated CFs (midpoint and endpoint) are available as spreadsheet tables in the Supplement B and are mapped in the Supplement C. Fig. 3 plots the distribution of the municipality scale midpoint CFs (boxplots) and the values of the country-scale midpoint CFs (green dots). At both scales, we observed that the expansion of most land covers is detrimental to the majority of ecosystem functions. The few exceptions are the beneficial effects of expanding cropland on crop production, of expanding grassland on livestock production, and of expanding forest on carbon sequestration, pollination and apple production. We also found that cropland expansion can potentially improve pollination and apple production in some municipalities. However, it is still detrimental at the national scale (average effect over the country).

Considering the distribution of the CFs, we found, for the country-scale CFs (green dots in Fig. 3), that the ratio between the least and the most impacting land cover types (considering only negative CFs) is equal to 1.5 for crop production, between 4 and 5 for carbon sequestration, pollination by *Osmia*, and the production of apples and livestock, and is equal to 7 for pollination by *Bombus*. For the municipality-scale CFs, the ratio between the least and most impacting municipalities goes up to thousands for some CFs (considering all possible combinations of land cover and function). Therefore, the CFs allow to show that the different land covers can have different impacts on the ecosystem functions and among the municipalities.

Finally, considering potential interactions among ecosystem functions, we mapped as an example in Fig. 4, the municipality-scale CFs that assess the impacts of expanding urban land on pollination by *Osmia* and the related production of apples (this second depends on the first). There, we observed that the impacts of urban expansion on pollination and apple production are not correlated among the

municipalities. The statistical analysis shows that there is a weak, positive correlation that is not significant: Spearman's  $\rho = 0.15$ , with  $p = 0.15$ . In some municipalities, the impact on pollination is among the highest, while the impact on apple production is the lowest, and vice-versa. We explain the reasons for these findings in the "Discussion" section.

### 3.2. Endpoint characterization factors for Luxembourg

The country-scale endpoint CFs, which we calculated using the midpoint and mid-to-endpoint CFs (cf. the Supplement A.1), are plotted in Fig. 5. We found that land use shall, in most cases, induce a loss of ecosystem services to society, through the opportunity costs associated with the loss of marketed ecosystem products (market value in Fig. 5-B) and through the social costs of a diminished carbon sequestration (climate regulation in Fig. 5-A). Only the expansion of cropland and grassland might generate further market benefits (this is not the case for all municipalities – cf. the Supplement C), and only forest expansion may improve climate regulation.

Furthermore, we observed that among land cover types, the potential impacts of land use on both indicators vary from positive to negative values and are quite homogeneously distributed between their minimum and maximum values. Therefore, the endpoint CFs, like the midpoint CFs, allow us to illustrate differences between the land cover types. We also noticed that when impacts on the market value increase, impacts on climate regulation decrease (this can be observed along land cover types from the left to the right in Fig. 5-A and B).

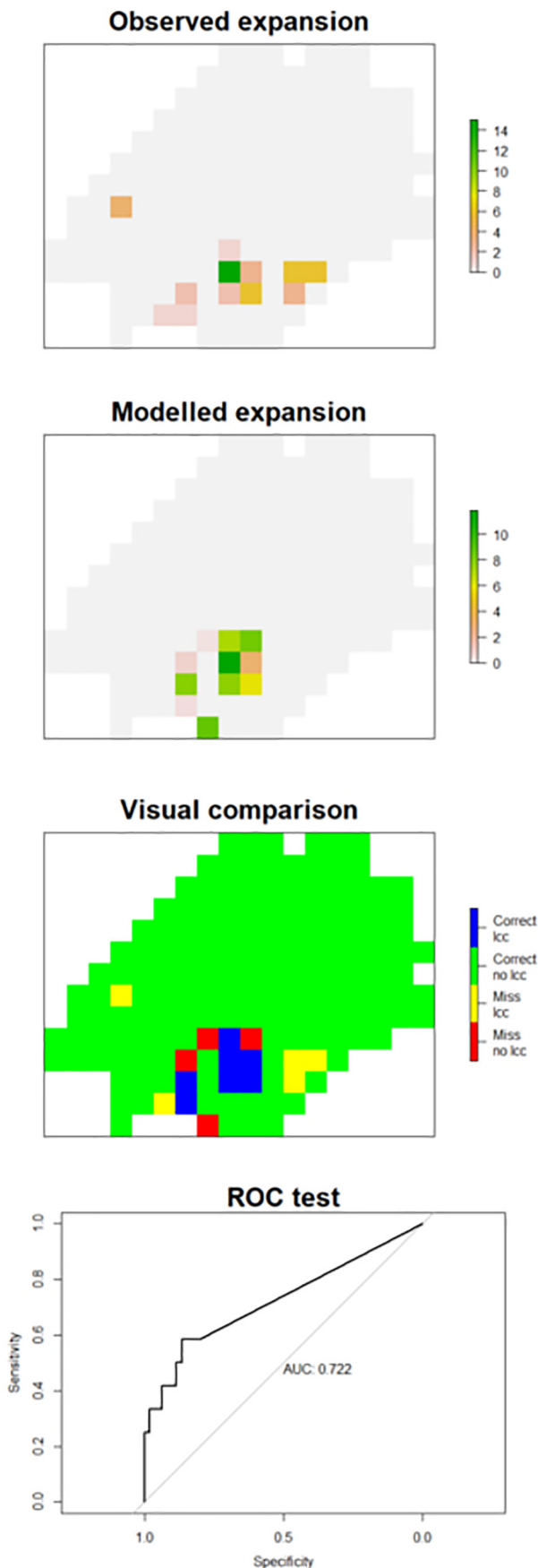
In Fig. 5-C, we decomposed the value of the endpoint CFs that assess impacts on the market value of ecosystem services. We observed that changes in the value of apples are much smaller than changes in the value of crops and livestock (lower by two orders of magnitude) and represent <0.1% of the final values of the CFs.

Finally, the municipality scale endpoint CFs vary more or less across municipalities, like at the midpoint level. As an example, we mapped in Fig. 6 the municipality scale endpoint CFs that we calculated for urban expansion. On the one hand, impacts on market production do not substantially vary: the ratio between the highest and the lowest CFs is below 2. On the other hand, impacts on climate regulation have a wider range: removing the CFs equal to 0, the ratio between the highest and lowest CFs is superior to 4. Also, in this case of urban expansion, we observed a synergy between both ecosystem services. The impacts are significantly and positively correlated: Spearman's  $\rho = 0.85$ , with  $p < 0.001$  (approximate  $p$  because there are ties). Urban land expands in place of cropland and grassland in our model, with a different ratio of converted cropland to converted grassland per municipality. Since grassland sequesters carbon while cropland does not, and since grassland provides a higher market value per  $m^2$  than cropland, both ecosystem services are more impacted in the municipalities where urban land expands more over grassland. Different synergies and trade-offs can be observed for the other types of land cover (cf. the Supplement C).

### 3.3. Validity of the models

The results of the land use model's validation through the ROC method are summarized in Fig. 7. There, we plotted, per land cover type, the area under the curve value (theoretically between 0 and 1) that we obtained for all the municipalities. As can be seen, the accuracy of the model's predictions depends on the land cover type. Overall, it better simulates the changes in the built land covers (urban, industry and infrastructure) than the changes in cropland, grassland and forest. In a few municipalities, the model is worse than a random model. In these cases, we did not calculate CFs. In general, the model would need further improvement and validation for actual policy decision. However, as proof-of-principle, we have demonstrated that the set-up works, i.e., the modeling prototype performs better than a random





model. We thus concluded that the model is useful to display the advantages of our new calculation approach.

Concerning the validation of BIOME-BGC, reported in Fig. 8, we observed that this submodel correctly simulates the daily gross primary productivity for the three tested species. It is a bit low in the case of *Fagus*, which we partly explain by the fact that this species is compared with results from a mixed forest (cf. the Supplement A.4.3.2). Still, we concluded that this part of the model is valid enough for our purpose. Note also that we considered annual aggregates in our model, which are less sensitive to daily punctual errors. For the year 2012, errors in the annual aggregates are below 10%.

Finally, InVEST pollination simulates total yields of 2669 tons of apples in 2006 and of 2527 tons in 2012. For the observed yields, 2515 tons were produced in 2007, 2406 tons in 2010 and 2419 tons in 2015 (these are the closest available years from STATEC, 2019). Therefore, the simulated yields are higher than the observations by approximately 5%. These were the best results that we could obtain through the calibration of the model. Again, we could assume that this error is acceptable in our case.

#### 4. Discussion

##### 4.1. The advantages of the proposed approach

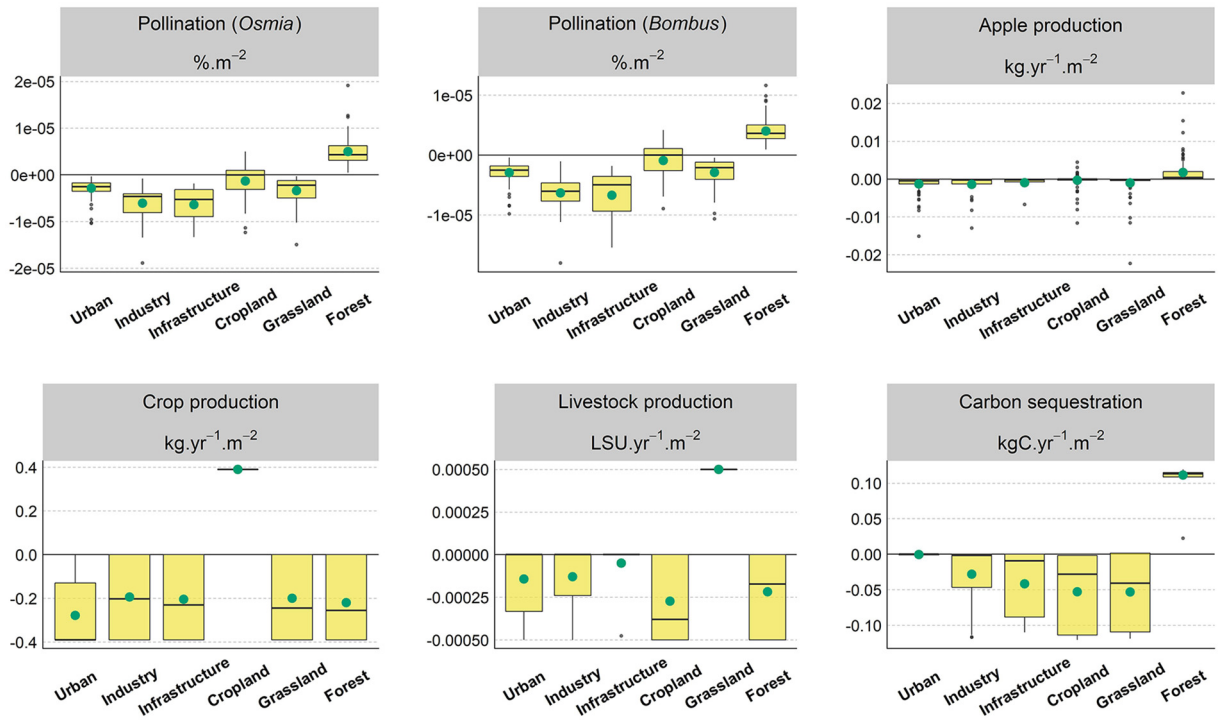
Using an integrated model of land cover and ecosystem services for calculating CFs allowed us to highlight differences between the impacts of expanding different land cover types. This is what is expected from CFs in order, for instance, to compare production systems with regard to their impacts on ecosystem services using LCA.

Into details, our approach allows to calculate CFs that assess the trade-offs induced by land cover changes among ecosystem functions and services. This is quite evident in the results of Fig. 5, where land use clearly induces a trade-off between the provision of climate regulation and of marketed ecosystem products. Such results show the limited capacity of land and ecosystems to simultaneously provide multiple ecosystem services at their maximum potential, as already illustrated by Foley et al. (2005) and observed by Holt et al. (2016) and Jopke et al. (2015) for instance.

In addition, thanks to the integrated nature of the model, we could consider interactions among ecosystem functions, since apple production depends on pollination in our model (in the structure of the InVEST model).

Moreover, we could encompass the effects of changing land cover patterns on ecosystem functions in the calculation of CFs, i.e., the landscape effects (Bennett and Isaacs, 2014; Pufal et al., 2017; Ricketts et al., 2008). This can be seen in the results of Fig. 4 where impacts on pollination and apple production are not correlated, although the latter depends on the former. That is because a land cover change can affect pollination without affecting apple production, when there are no orchards close to where the change takes place, and vice-versa. The municipalities where apple production is mostly affected are those where orchards are close to existing urban centers, from where urban land tends to expand. In this line, at the country scale (Fig. 3), we observe that the expansion of infrastructures is the most detrimental to pollination, while the expansion of industry is the most detrimental to apple production. This is because infrastructure may replace forests, which are an important habitat for pollinators, while industry takes place around urban centers where orchards are most present. Therefore, all these impacts – i.e., the values of the CFs – do depend on the conformation of land cover patches and how this changes.

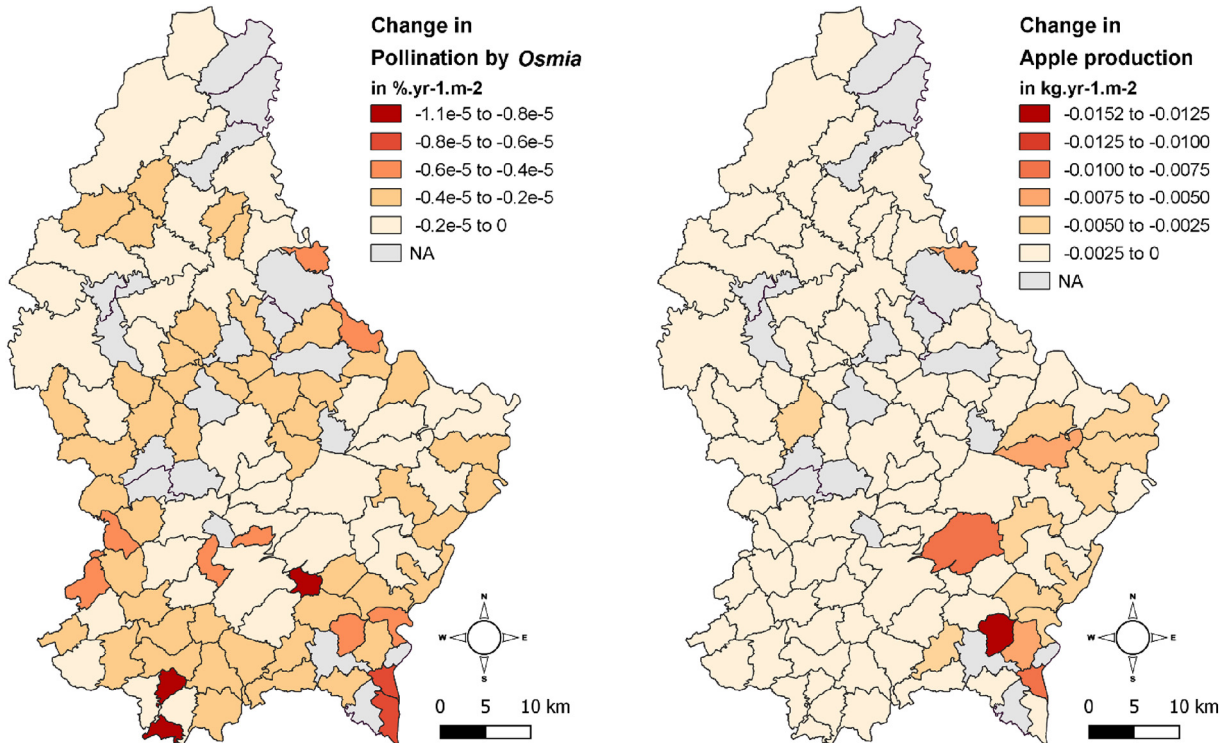
**Fig. 2.** Method to validate the land use model. The validity of the model, here in the case of the expansion of infrastructure in a municipality, is tested through the comparison for a past period (from 2000 to 2012) of the observed (top left) and predicted (top right) land cover changes. This comparison is summarized through a map (bottom left) and a ROC curve (bottom right).



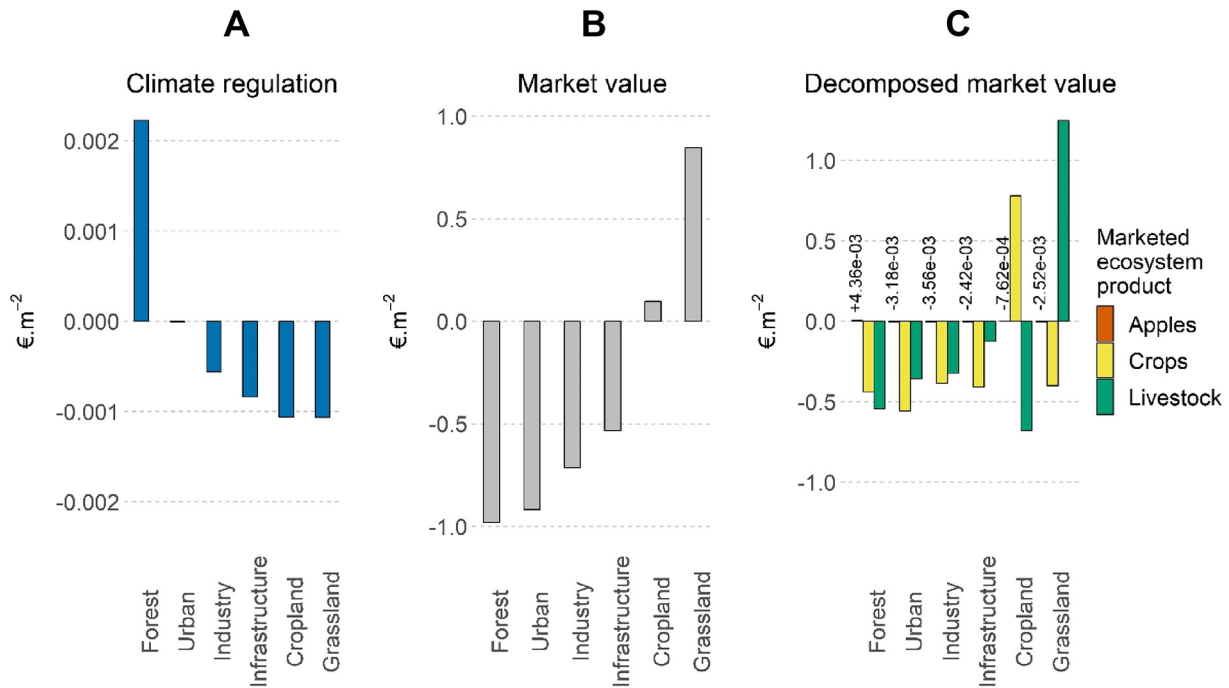
**Fig. 3.** Midpoint characterization factors (vertical axis – the unit of the CFs are in the grey box title) per land cover type (horizontal axis category) and impacted ecosystem function (panel category). Each boxplot displays the distribution of the regionalized CFs over the 116 municipalities of Luxembourg. In a boxplot, the bold line inside the box indicates the median of the plotted distribution, while the two sides of the box indicate its lower and upper quartiles. Also, by  $x$  the interquartile distance, the outside bars (“whiskers”) indicate the values within  $1.5 \times x$  from the quartiles. Finally, the points represent outliers, which are more than  $1.5 \times x$  away from the quartiles. The green dots represent the country-scale CFs, which are calculated as arithmetic means of the municipality scale factors. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Finally, we could also take into account legal aspects in the calculation of our CFs, namely protected areas. This can be observed in the municipalities where cropland expansion improves pollination, as we

pointed out (cf. Fig. 3). In these municipalities, all forest areas are protected, while in others, only parts of them are protected. Hence, cropland expands only over grassland in those areas. Since grassland



**Fig. 4.** Maps of municipality-scale midpoint characterization factors to assess the impacts of expanding urban land on pollination by *Osmia* (left) and apple production (right). Since urban expansion was not modeled in the “NA” municipalities where no past expansions were observed (cf. Supplement 4.2.2), characterization factors were not calculated for these regions.

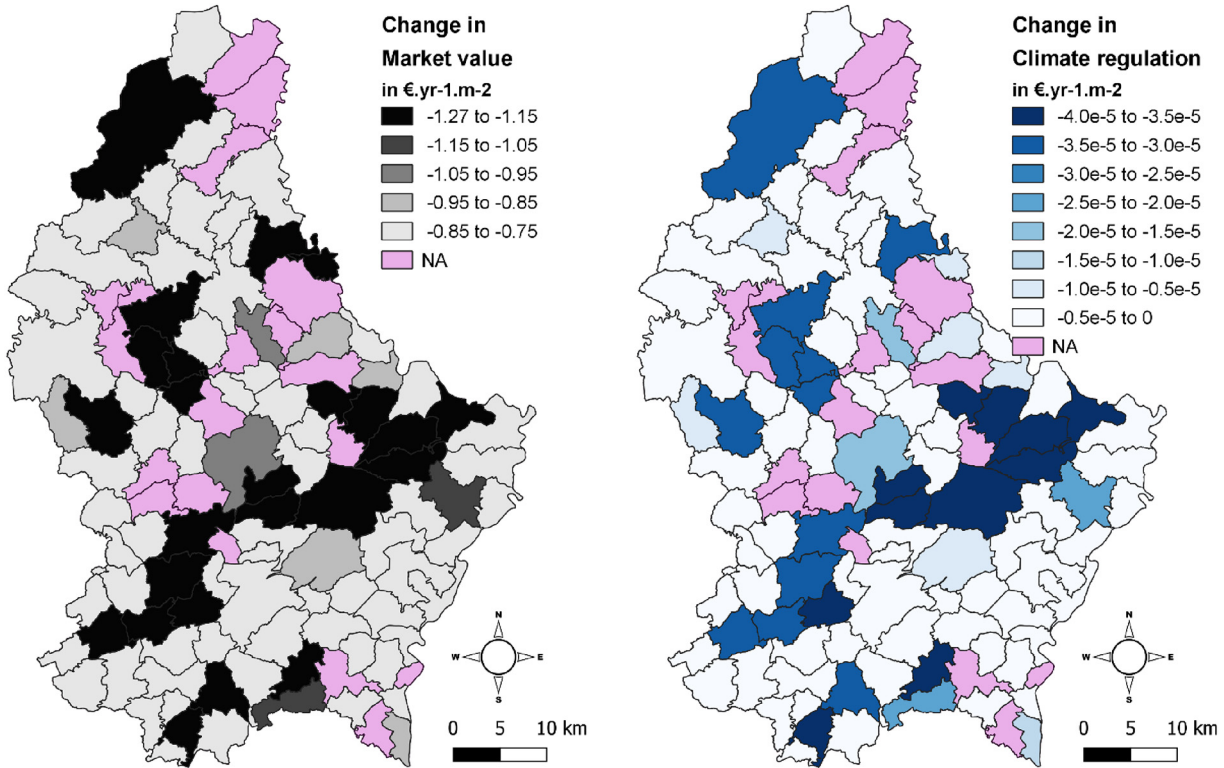


**Fig. 5.** Endpoint characterization factors (CFs) per land cover type for impacts on two ecosystem services: climate regulation (A) and market value (B). Market value is equal to the value of apples, crops and livestock (decomposition in C, where the values of apples are labeled for visualization).

is a less suitable habitat for pollinators than cropland in our implementation of the InVEST model – most of the grassland is dedicated to grazing in Luxembourg (cf. the Supplement A.4.3.1) –, the expansion of cropland has a positive effect on pollination in these municipalities.

Overall, these aspects (trade-offs between ecosystem services, interactions between ecosystem functions, landscape effects and socio-

economic characteristics) were still not addressed in the calculation of CFs (Arbault et al., 2014; Maia de Souza et al., 2018; Othoniel et al., 2016; Teixeira, 2014). Addressing them, our approach for calculating CFs better complies with the state-of-the-art assessment of land use and impacts on ecosystem services (e.g., Yan, 2018). In this regard, we calculated the first CFs that allow to assess impacts on pollination (we



**Fig. 6.** Maps of municipality scale endpoint characterizations factors to assess the impacts of expanding urban land on the market value (left) and the climate regulation value (right) of ecosystem services. Since urban expansion was not modeled in the “NA” municipalities where no past expansions were observed, characterization factors were not calculated for these regions.

Validation of the land use model:  
Area under the curve (ROC test) per land cover and over municipalities

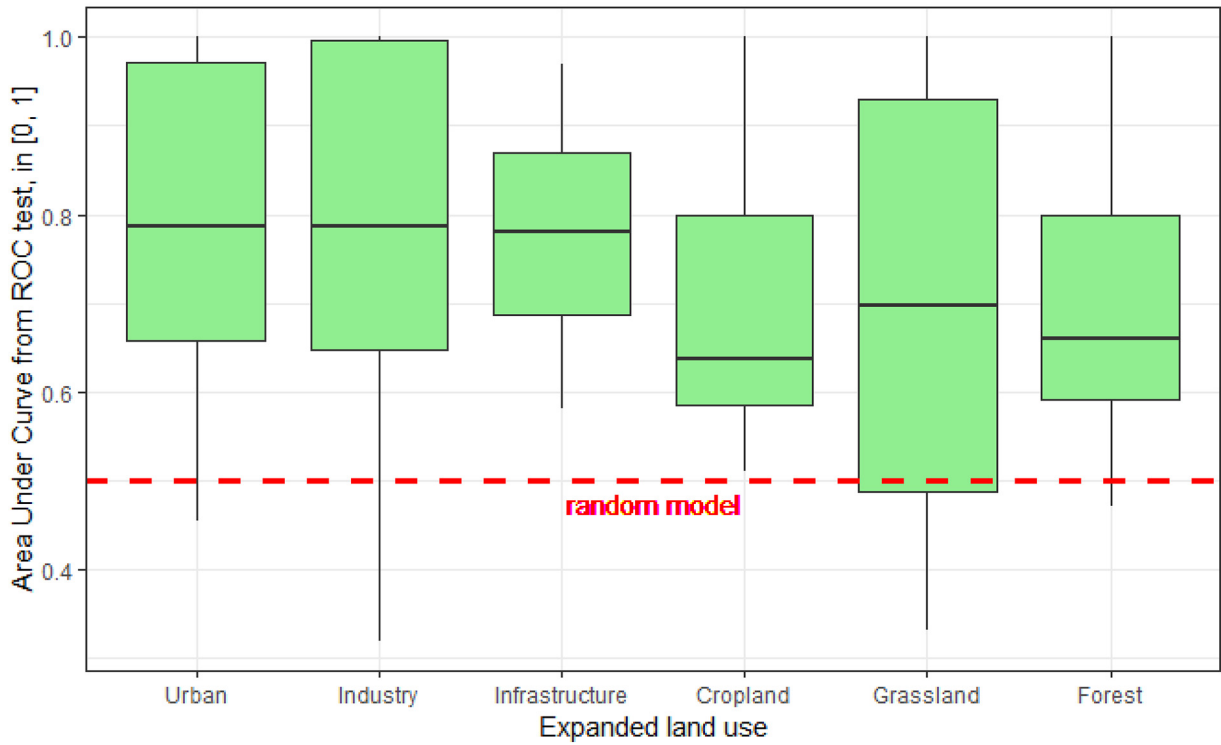


Fig. 7. Results of the land use model's validation with a ROC test (summary statistics: the area under the curve in [0,1], with 1 meaning that the model's prediction is perfect and with 0.5 being the validity of a random model). For each type of land cover, a boxplot describes the distribution of the model's validity over the municipalities of Luxembourg. In a boxplot, the bold line inside the box indicates the median of the plotted distribution, while the two sides of the box indicate its lower and upper quartiles. Also, be  $x$  the interquartile distance, the outside bars ("whiskers") indicate the values within  $1.5 \times x$  from the quartiles. Finally, the points represent outliers, which are more than  $1.5 \times x$  away from the quartiles.

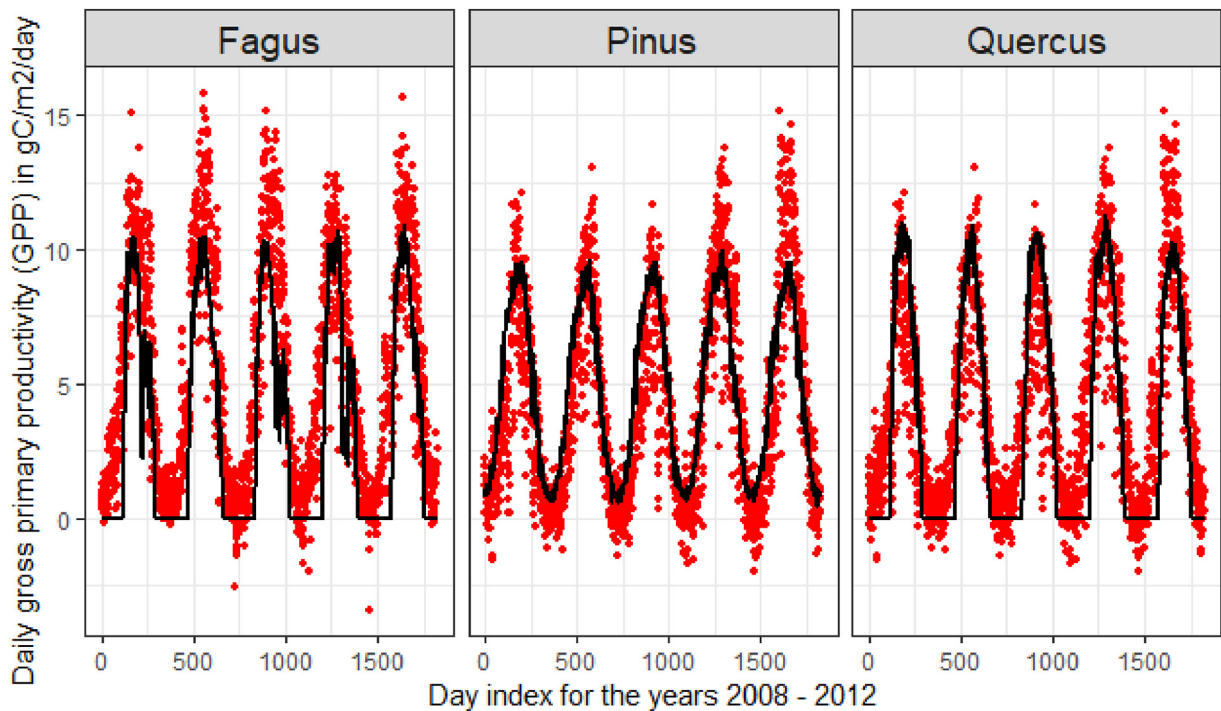


Fig. 8. Comparison of the carbon flux simulated by BIOME-BGC (black lines) with Eddy-covariance measures (red dots; from FLUXNET, 2018). As can be seen, the model replicates well the observed data. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

rely on Crenna et al., 2017, for this assertion). We argue that we could not have achieved this 1) if we had followed the framework of the UNEP/SETAC, and 2) without the initial assumption of marginal demands for land covers. We discuss these two arguments in the section "Towards a real-world application of the approach".

Considering the regionalization of the CFs, this provides insights on the spatial variability of impacts on ecosystem services, in our case, among the municipalities of Luxembourg. As can be seen in Fig. 3, this variability depends on the types of the assessed land cover and ecosystem function/service. For instance, the impact of urban expansion on pollination varies more, over the municipalities, than the impact of urban expansion on crop production. This implies that the impacts calculated in LCA thanks to country scale CFs could be quite far from the actual impacts. Therefore, municipality scale CFs should be used when it is relevant from a decision-making point of view. That is when the decision-makers using LCA can decide on where to implement their production and can conduct an inventory of land use at such a local scale. This will generally be the case when modeling the "foreground system" of the studied product's life cycle (Hauschild et al., 2018). Furthermore, the high variability should be communicated with the country-scale CFs, as discussed in the following section.

#### 4.2. Building the credibility of the characterization factors

Because we propose to use quite complex model in a predictive way to calculate CFs, their values may seem "unrealistic" at first glance. Therefore, it is necessary to establish their credibility, in order to foster their integration in decision-making practices. To this end, several efforts are required.

First, the calculation of CFs should be transparent. That is why we thoroughly described our approach, so that other practitioners can replicate it in a standardized way.

Second, the validity of the characterization model should be communicated, as a proof of its predictive capacity (Pontius and Parmentier, 2014). In our case, the part of the characterization model that simulates land use is the least valid one, while the other parts appear as more accurate. The validity of our land use submodel is a critical issue that hampers the direct application of the CFs calculated in this paper. However, we argue that this does not question the proposed calculation approach and the use of such model for calculating CFs. Instead, we argue that improving the modeling of land use and land cover changes should be a priority for calculating credible CFs. Into details, the land use submodel performs worse for cropland, grassland and forests (cf. Fig. 7). To improve the modeling of cropland and grassland, the granularity of these land cover classes could be increased, i.e., cropland could be divided into rapeseed, maize, wheat, etc., and grassland into natural grasslands, pastures and meadows. Also, an economic model could be added, e.g., a partial equilibrium model for the agricultural sector of Luxembourg (Rege et al., 2016). As a result, the land use model should better simulate the land covers that are dedicated to agricultural uses. For simulating the expansion of forests, we encountered the issue that this type of land cover did not substantially expand in the past. Therefore, we had little information available to calibrate the model. In fact, this remark applies to all land uses in Luxembourg. Because relatively few land use changes occurred in the past, there is a weak signal to interpret, which is always more difficult to simulate. On top of this, more accurate land use data could be used, for instance from direct observations (Ministère de la mobilité et des travaux publics du Grand-Duché du Luxembourg, 2019). In this study, we relied on CORINE land cover data because we needed at least three points in time to calibrate and validate the model (only two were available for direct observations). Otherwise, other models than those adopted in this study should be tested, as they may be more accurate and practical for calculating CFs. In this regard, comparable standards of validity should be used in order to benchmark the quality of CFs calculated with different models or in different regions. Nonetheless, which approach to follow for the

validation of spatial land cover models, and how to benchmark validation results between different models and applications are still debated questions (Pontius and Millones, 2011; van Vliet et al., 2016).

Finally, the quality of the CFs should be better communicated, for example by presenting confidence intervals that quantify the CF's potential error and uncertainty (Baustert et al., 2018). In the present study, we did not calculate such intervals, since we considered the model as not valid enough to disseminate our CFs to decision-makers. Still, we provide a few insights on how to calculate those intervals. Concerning the evaluation of the CFs' error, this should be related to the validation of the models and should depend on the observed accuracy over the validation period. Concerning the assessment of the CFs' uncertainty, this may require dedicated efforts and potentially new methodological developments, given the current state of uncertainty analysis practices for integrated assessment models (Baustert et al., 2018). Following a precautionary principle, we qualitatively describe the level of uncertainty of our CFs as "very uncertain" and do not recommend their direct use in LCA. We identify the potential sources of uncertainty in the Supplement A.4.6. One source that is specific to our approach is the calculation of country-scale CFs from municipality-scale CFs. In the present article, we calculated the former as the arithmetic mean of the latter. However, it would also make sense to weight the municipality-scale CFs before computing their mean. For instance, for a given land cover type  $s$ , it would make sense to assign to each municipality  $r$  a weight that is equal to the ratio of the past expansion of  $s$  in  $r$  over the past expansion of  $s$  in the whole country. Nonetheless, this raises further questions about, for example, the period over which past expansions should be observed.

#### 4.3. Towards a real-world application of the approach

In the present study, we developed a novel approach for calculating CFs and tested it focusing on the case of Luxembourg. Since our approach does provide several advantages, we hope that it will be further applied, using improved versions of our models or other models, to calculate CFs for all the countries in the world. Accordingly, to help the future calculators of CFs, we identify three challenges that should be tackled through such endeavor.

First, calculating CFs according to our approach for many countries, if not all, could be time consuming. Nevertheless, an automatized procedure could be established in order to optimize the efforts invested in the implementation of characterization models that cover the world (e.g., Martínez-López et al., 2018). Also, the analysis of CFs calculated in different countries could focus on identifying national characteristics that influence the variability of the CFs. Understanding what causes the CFs to vary between countries could help reduce the calculation of CFs for the whole world, since the CFs calculated for a country may be used as a proxy for other countries with similar socio-economic characteristics and land use change constraints. As an example, CFs calculated for Luxembourg may be valid for the entire Benelux region, since Belgium, the Netherlands and Luxembourg share common characteristics in terms of ecological conditions, land and ecosystems management, territorial organization and agricultural development. Still, this requires a validation.

Second, the list of land covers, ecosystem functions and services for which CFs are calculated should be increased. Since these issues were already analyzed by other research, notably in Koellner et al. (2013a), Maia de Souza et al. (2018) and Alejandre et al. (2019), we discuss them in the Supplement A.3. Still, a specific point on which our results bring insights is the valuation of the benefits provided by ecosystem services. In the present study, we valued them in monetary terms. Such monetization of ecosystem services is still debated (Saarikoski et al., 2016), mostly because it can hardly encompass the values of the services supported by non-marketed ecosystem products (Polasky et al., 2015). Moreover, monetary valuation can become counter-productive if it induces a commodification of ecosystem services (Gómez-

Baggethun et al., 2010; Gómez-Baggethun and Ruiz-Pérez, 2011). Considering our findings, we see, in the case of forest, that its expansion can improve pollination and therefore the production of apples (cf. Fig. 3), which are marketed. However, because forests spread over cropland and grassland in our model, the benefits underpinning the production of apples are cancelled by the potential losses of marketed crops and livestock. As a result, forest expansion appears to have an overall negative impact on the production of ecosystem products. While this may be true from an economic point of view, such aggregation can hide details that may be relevant for decision makers if, for instance, they strongly prefer apples to crops and livestock for non-economic reasons. Therefore, it seems advisable to consider midpoint and endpoint impacts in parallel when ecosystem services are monetized at the endpoint level. Furthermore, non-monetary valuation methods could also be used in endpoint characterization models. The DALYs indicator, for instance, which is used in LCA to assess impacts on health, could serve to value the services that affect people's health for example (Pfister et al., 2009, 2014). Moreover, Pascual et al. (2017) recently proposed an approach to value nature's contribution to people (ecosystem services in other words) in the frame of the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES). They argue, like Jacobs et al. (2016) for instance, in favor of developing decision-support tools that provide multiple value indicators (and therefore use several valuation methods) in order to acknowledge the intrinsic, instrumental and relational values of ecosystem services. Our approach would certainly benefit from using other valuation methods. For instance, we did not account for an important social benefit obtained from apples, which are much used in Luxembourg for the non-marketed production of traditional liquors that eventually are a non-negligible ecosystem service.

In the end, our CFs are not compatible with the current modeling of land use in LCI databases. According to the framework of the UNEP-SETAC (Koellner et al., 2013b; Milà et al., 2007), the land use associated with a unit process is modeled with three types of stressors in LCI databases. First, a "land occupation" stressor that represents keeping land under a certain type of land use/cover for a certain period. Second and third, two "land transformation" stressors that are called "from" and "to", and that represent the conversion of land "from" a certain type "to" another. The impact of each of these stressors should then be assessed with a specific CF. These steps are further detailed in the Supplement D. Therefore, according to this framework, we should calculate CFs that assess the impact on ecosystem services of expanding cropland, for instance, without considering in place of what it expands. Inversely, we should calculate CFs that assess the impact on ecosystem services of converting a forest, for instance, without considering what replaces it. In this, we see several issues, which we detail in the Supplement D. In summary, the main issue comes from the fact that the impact of expanding a given land cover depends on where it occurs and what it replaces, i.e., there is a landscape effect. Furthermore, where this land cover expands and what it replaces also depend on each other. Therefore, when modeling them separately, which the framework of the UNEP-SETAC implies, all these interactions are omitted. As a consequence, we argue that the modeling of land use in LCI could be rethought in order to fit our initial (and ideally more representative) assumption of quantifying "demands" for land covers as drivers of land cover changes (cf. Chaplin-Kramer et al. (2017) too). This would make the land use results of LCI more in line with the structure of existing land use models that generally aim to simulate where and in place of what land covers will increase in response to driving land demands (Verburg et al., 2006). Subsequently, this would allow to use existing models of ecosystem functions and services that aim, in part, to simulate landscape effects. We thus propose to inventory, for all the unit processes in LCI databases, land use stressors that would describe by how much different land covers will be expanded to conduct these processes. These stressors should be close to the current transformation "to" stressors. For instance, producing  $y$  kg of maize will imply a demand for  $x$  m<sup>2</sup> of

cropland. The impacts on ecosystem services of these stressors would then be assessed according to Eq. 4 with CFs like ours. The duration of land use, if needed, could be expressed with a specific variable, named "duration of land use", for instance. Otherwise, demands for land covers could be expressed in m<sup>2</sup>.yr, like the current "land occupation" stressors, in order to calculate long-term impacts of land use. Ultimately, such an approach deserves further research since it would allow a more representative assessment of the impacts of land use on ecosystem services in LCA.

## 5. Conclusions

In this paper, we developed, for the case of Luxembourg, an original method and related modeling framework for characterizing the impacts of land use on ecosystem services in LCA. Accordingly, we could overcome several limitations that were previously observed in the calculation of CFs dedicated to these impacts. A main reason is that we used models that are acknowledged in the field of ecosystem services assessment, and so far considered outside of the LCA research domain. Because these models can be quite complex and uncertain, their validity must be evaluated in order to establish the credibility of the CFs. Once this is done, we can benefit from the fact that they have been sharpened through past research in the field of integrated ecosystems modeling, and thus accurately simulate the cause effect chain underlying the impact of land use on ecosystem services. Furthermore, using models that are already established in decision practices allows to calculate CFs that yield LCA impact indicators that are well-anchored in such decision practices.

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## Appendix A. Supplementary data

Detailed formulas for the calculation of midpoint and endpoint CFs, information on the midpoint characterization model, and other details can be found in the Supplement A. The values of the characterization factors are reported in the Supplement B. They are mapped in the Supplement C, together with the simulations that served to calculate the CFs. We further discuss the inventory of land use in the Supplement D. Supplementary data to this article can be found online at doi: <https://doi.org/10.1016/j.scitotenv.2019.07.180>.

## References

- Administration du Cadastre et de la Topographie, 2007. Geoportail.lu: the official online platform of the Luxembourgish government for the collection and distribution of spatial datasets. <https://map.geoportail.lu/>.
- Alejandre, E.M., van Bodegom, P.M., Guinée, J.B., 2019. Towards an optimal coverage of ecosystem services in LCA. *J. Clean. Prod.* 231, 714–722. <https://doi.org/10.1016/j.jclepro.2019.05.284>.
- Arbault, D., Rivière, M., Rugani, B., Benetto, E., Tiruta-Barna, L., 2014. Integrated earth system dynamic modeling for life cycle impact assessment of ecosystem services. *Sci. Total Environ.* 472, 262–272. <https://doi.org/10.1016/j.scitotenv.2013.10.099>.

- Bakshi, B.R., Ziv, G., Lepech, M.D., 2015. Techno-ecological synergy: a framework for sustainable engineering. *Environ. Sci. Technol.* 49, 1752–1760. <https://doi.org/10.1021/es5041442>.
- Balvanera, P., Quijas, S., Karp, D.S., Ash, N., Bennett, E.M., Boumans, R., Brown, C., Chan, K.M.A., Chaplin-Kramer, R., Halpern, B.S., Honey-Rosés, J., Kim, C.-K., Cramer, W., Martínez-Harms, M.J., Mooney, H., Mwampamba, T., Nel, J., Polasky, S., Reyers, B., Roman, J., Turner, W., Scholes, R.J., Tallis, H., Thonike, K., Villa, F., Walpole, M., Walz, A., 2017. Ecosystem services. *The GEO Handbook on Biodiversity Observation Networks*. Springer International Publishing, Cham, pp. 39–78. [https://doi.org/10.1007/978-3-319-27288-7\\_3](https://doi.org/10.1007/978-3-319-27288-7_3).
- Bare, J., 2011. Recommendation for land use impact assessment: first steps into framework, theory, and implementation. *Clean Techn. Environ. Policy* 13, 7–18. <https://doi.org/10.1007/s10098-010-0290-8>.
- Baude, M., Meyer, B.C., Schindewolf, M., 2019. Land use change in an agricultural landscape causing degradation of soil based ecosystem services. *Sci. Total Environ.* 659, 1526–1536. <https://doi.org/10.1016/j.scitotenv.2018.12.455>.
- Baustert, P., Othoniel, B., Rugani, B., Leopold, U., 2018. Uncertainty analysis in integrated environmental models for ecosystem service assessments: frameworks, challenges and gaps. *Ecosyst. Serv.* 33, 110–123. <https://doi.org/10.1016/j.ecoser.2018.08.007>.
- Becerra-Jurado, G., Philipsen, C., Kleeschulte, S., 2015. Mapping and assessing ecosystems and their services in Luxembourg - Final Synthesis Report. *space4environment sàrl; Ministère du développement durable et des infrastructures (MDDI) / Département de l'environnement, Luxembourg*.
- Bennett, A.B., Isaacs, R., 2014. Landscape composition influences pollinators and pollination services in perennial biofuel plantings. *Agric. Ecosyst. Environ.* 193, 1–8. <https://doi.org/10.1016/j.agee.2014.04.016>.
- Blanco, C.F., Marques, A., van Bodegom, P.M., 2018. An integrated framework to assess impacts on ecosystem services in LCA demonstrated by a case study of mining in Chile. *Ecosyst. Serv.* 30, 211–219. <https://doi.org/10.1016/j.ecoser.2017.11.011>.
- Boumans, R.M.J., Roman, J., Altman, I., Kaufman, L., 2015. The multiscale integrated model of ecosystem services (MIMES): simulating the interactions of coupled human and natural systems. *Ecosyst. Serv.* 12, 30–41. <https://doi.org/10.1016/j.ecoser.2015.01.004>.
- Brandão, M., I Canals, L.M., 2013. Global characterisation factors to assess land use impacts on biotic production. *Int. J. Life Cycle Assess.* 18, 1243–1252. <https://doi.org/10.1007/s11367-012-0381-3>.
- Butchart, S.H.M., Walpole, M., Collen, B., Van Strien, A., Scharlemann, J.P.W., Almond, R.E.A., Baillie, J.E.M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K.E., Carr, G.M., Chanson, J., Chenery, A.M., Csirke, J., Davidson, N.C., Dentener, F., Foster, M., Galli, A., Galloway, J.N., Genovesi, P., Gregory, R.D., Hockings, M., Kapos, V., Lamarque, J.F., Leverington, F., Loh, J., McGeoch, M.A., McRae, L., Minasyan, A., Morcillo, M.H., Oldfield, T.E.E., Pauly, D., Quader, S., Revenga, C., Sauer, J.R., Skolnik, B., Spear, D., Stanwell-Smith, D., Stuart, S.N., Symes, A., Tierney, M., Tyrrell, T.D., Vié, J.C., Watson, R., 2010. Global biodiversity: indicators of recent declines. *Science* (80-) 328, 1164–1168. <https://doi.org/10.1126/science.1187512>.
- Cao, V., Margni, M., Favis, B.D., Deschênes, L., 2015. Aggregated indicator to assess land use impacts in life cycle assessment (LCA) based on the economic value of ecosystem services. *J. Clean. Prod.* 94, 56–66. <https://doi.org/10.1016/j.jclepro.2015.01.041>.
- Chaplin-Kramer, R., Sim, S., Hamel, P., Bryant, B., Noe, R., Mueller, C., Rigarsford, G., Kulak, M., Kowal, V., Sharp, R., Clavreul, J., Price, E., Polasky, S., Ruckelshaus, M., Daily, G.C., 2017. Life cycle assessment needs predictive spatial modelling for biodiversity and ecosystem services. *Nat. Commun.* 8, 15065. <https://doi.org/10.1038/ncomms15065>.
- Chaudhary, A., Carrasco, L.R., Kastner, T., 2017. Linking national wood consumption with global biodiversity and ecosystem service losses. *Sci. Total Environ.* 586, 985–994. <https://doi.org/10.1016/j.scitotenv.2017.02.078>.
- Clermont, A., Eickermann, M., Kraus, F., Hoffmann, L., Beyer, M., 2015. Correlations between land covers and honey bee colony losses in a country with industrialized and rural regions. *Sci. Total Environ.* 532, 1–13. <https://doi.org/10.1016/j.scitotenv.2015.05.128>.
- Costanza, R., de Groot, R.S., Sutton, P., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S., Turner, R.K., 2014. Changes in the global value of ecosystem services. *Glob. Environ. Chang.* 26, 152–158. <https://doi.org/10.1016/j.gloenvcha.2014.04.002>.
- Costanza, R., de Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., Farber, S., Grasso, M., 2017. Twenty years of ecosystem services: how far have we come and how far do we still need to go? *Ecosyst. Serv.* 28, 1–16. <https://doi.org/10.1016/j.ecoser.2017.09.008>.
- Crenna, E., Polce, C., Collina, E., 2017. Pollinators in life cycle assessment: towards a framework for impact assessment. *J. Clean. Prod.* 140, 525–536. <https://doi.org/10.1016/j.jclepro.2016.02.058>.
- De Baan, L., Mutel, C.L., Curran, M., Hellweg, S., Koellner, T., 2013. Land use in life cycle assessment: global characterization factors based on regional and global potential species extinction. *Environ. Sci. Technol.* 47, 9281–9290. <https://doi.org/10.1021/es400592q>.
- Egoh, B., Drakou, E.G., Dunbar, M.B., Maes, J., Willemens, L., Drakou, E.G., Dunbar, M.B., Maes, J., 2012. Indicators for Mapping Ecosystem Services: A Review. <https://doi.org/10.2788/41823>.
- European Environment Agency, 2015. European ecosystem assessment – Concept, data and implementation. EEA Technical Report <https://doi.org/10.2800/629258>.
- European Environment Agency, 2019. Copernicus Land Monitoring Service - Corine Land Cover. <https://land.copernicus.eu/pan-european/corine-land-cover>.
- FLUXNET, 2018. See the acknowledgements. <https://fluxnet.fluxdata.org/>.
- Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin III, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. *Science* 309, 570–574. <https://doi.org/10.1126/science.1111772>.
- Frischknecht, R., Joliet, O., 2016. Global guidance for life cycle impact assessment indicators - Volume 1. UNEP/SETAC Life Cycle Initiative. <https://www.lifecycleinitiative.org/training-resources/global-guidance-lcia-indicators-v-1/>.
- Gómez-Baggethun, E., Ruiz-Pérez, M., 2011. Economic valuation and the commodification of ecosystem services. *Prog. Phys. Geogr.* 35, 613–628. <https://doi.org/10.1177/0309133311421708>.
- Gómez-Baggethun, E., de Groot, R.S., Lomas, P.L., Montes, C., 2010. The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. *Ecol. Econ.* 69, 1209–1218. <https://doi.org/10.1016/j.ecolecon.2009.11.007>.
- Greenleaf, S.S., Williams, N.M., Winfree, R., Kremen, C., 2007. Bee foraging ranges and their relationship to body size. *Oecologia* 153, 589–596. <https://doi.org/10.1007/s00442-007-0752-9>.
- Hauschild, M.Z., Huijbregts, M.A.J., 2015. Life Cycle Impact Assessment. Life Cycle Assessment: Principles and Practice. Springer International Publishing, Cham, pp. 46–53. <https://doi.org/10.1007/978-94-017-9744-3>.
- Hauschild, M.Z., Rosenbaum, R.K., Olsen, S.I., 2018. Life Cycle Assessment: Theory and Practice. Springer International Publishing, Cham. <https://doi.org/10.1007/978-3-319-56475-3>.
- Hauschild, M.Z., Rosenbaum, R.K., Olsen, S.I., 2017. Life Cycle Assessment: Theory and Practice, Life Cycle Assessment: Theory and Practice. Springer International Publishing, Cham. <https://doi.org/10.1007/978-3-319-56475-3>.
- Heijungs, R., Suh, S., 2002. *The Computational Structure of Life Cycle Assessment*. Springer, Netherlands.
- Hennani, R., 2017. *Horizon 2060: les frontaliers au Luxembourg* (No. 6).
- Holt, A.R., Alix, A., Thompson, A., Maltby, L., 2016. Food production, ecosystem services and biodiversity: we can't have it all everywhere. *Sci. Total Environ.* 573, 1422–1429. <https://doi.org/10.1016/j.scitotenv.2016.07.139>.
- Jacobs, S., Dendoncker, N., Martín-López, B., Barton, D.N., Gomez-Baggethun, E., Boeraeve, F., McGrath, F.L., Vierikko, K., Geneletti, D., Sevecke, K.J., Pipart, N., Primmer, E., Mederly, P., Schmidt, S., Aragão, A., Baral, H., Bark, R.H., Brieno, T., Brogna, D., Cabral, P., De Vreese, R., Lique, C., Mueller, H., Peh, K.S.-H., Phelan, A., Rincón, A.R., Rogers, S.H., Turkelboom, F., Van Reeth, W., van Zanten, B.T., Wam, H.K., Washbourn, C.L., 2016. A new valuation scheme: integrating diverse values of nature in resource and land use decisions. *Ecosyst. Serv.* 22, 213–220. <https://doi.org/10.1016/j.ecoser.2016.11.007>.
- Jeswani, H.K., Hellweg, S., Azapagic, A., 2018. Accounting for land use, biodiversity and ecosystem services in life cycle assessment: impacts of breakfast cereals. *Sci. Total Environ.* 645, 51–59. <https://doi.org/10.1016/j.scitotenv.2018.07.088>.
- Joliet, O., Frischknecht, R., Bare, J., Boulay, A.M., Bulle, C., Fantke, P., Gheewala, S., Hauschild, M.Z., Itsubo, N., Margni, M., McKone, T.E., Y Canals, L.M., Postuma, L., Prado-Lopez, V., Ridout, B., Sonnemann, G., Rosenbaum, R.K., Seager, T., Struijs, J., Van Zelm, R., Vigon, B., Weisbrod, A., 2014. Global guidance on environmental life cycle impact assessment indicators: findings of the scoping phase. *Int. J. Life Cycle Assess.* 19, 962–967. <https://doi.org/10.1007/s11367-014-0703-8>.
- Jopke, C., Kreyling, J., Maes, J., Koellner, T., 2015. Interactions among ecosystem services across Europe: Bagplots and cumulative correlation coefficients reveal synergies, trade-offs, and regional patterns. *Ecol. Indic.* 49, 46–52. <https://doi.org/10.1016/j.ecolind.2014.09.037>.
- Knudsen, M.T., Hermansen, J.E., Cederberg, C., Herzog, F., Vale, J., Jeanneret, P., Sarthou, J.P., Friedel, J.K., Balázs, K., Fjellstad, W., Kainz, M., Wolfrum, S., Dennis, P., 2017. Characterization factors for land use impacts on biodiversity in life cycle assessment based on direct measures of plant species richness in European farmland in the “Temperate Broadleaf and Mixed Forest” biome. *Sci. Total Environ.* 580, 358–366. <https://doi.org/10.1016/j.scitotenv.2016.11.172>.
- Koch, E.W., Barbier, E.B., Silliman, B.R., Reed, D.J., Perillo, G.M.E., Hacker, S.D., Granek, E.F., Primavera, J.H., Muthiga, N., Polasky, S., Halpern, B.S., Kennedy, C.J., Kappel, C.V., Wolanski, E., 2009. Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. *Front. Ecol. Environ.* 7, 29–37. <https://doi.org/10.1890/080126>.
- Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Goedkoop, M., Margni, M., i Canals, L.M., Müller-Wenk, R., Weidema, B.P., Wittstock, B., 2013a. Principles for life cycle inventories of land use on a global scale. *Int. J. Life Cycle Assess.* 18, 1203–1215. <https://doi.org/10.1007/s11367-012-0392-0>.
- Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., i Canals, L.M., Saad, R., de Souza, D.M., Müller-Wenk, R., 2013b. UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. *Int. J. Life Cycle Assess.* 18, 1188–1202. <https://doi.org/10.1007/s11367-013-0579-z>.
- Lambin, E.F., Meyfroidt, P., 2011. Global land use change, economic globalization, and the looming land scarcity. *Proc. Natl. Acad. Sci. U. S. A.* 108, 3465–3472. <https://doi.org/10.1073/pnas.1100480108>.
- Lindeijer, E., 2000. Review of land use impact methodologies. *J. Clean. Prod.* 8, 273–281. [https://doi.org/10.1016/S0959-6526\(00\)00024-X](https://doi.org/10.1016/S0959-6526(00)00024-X).
- Liu, X., Ziv, G., Bakshi, B.R., 2018. Ecosystem services in life cycle assessment - part 1: a computational framework. *J. Clean. Prod.* 197, 314–322. <https://doi.org/10.1016/j.jclepro.2018.06.164>.
- Lonsdorf, E., Kremen, C., Ricketts, T., Winfree, R., Williams, N.M., Greenleaf, S., 2009. Modelling pollination services across agricultural landscapes. *Ann. Bot.* 103, 1589–1600. <https://doi.org/10.1093/aob/mcp069>.
- Maia de Souza, D., Lopes, G.R., Hansson, J., Hansen, K., 2018. Ecosystem services in life cycle assessment: a synthesis of knowledge and recommendations for biofuels. *Ecosyst. Serv.* 30, 200–210. <https://doi.org/10.1016/j.ecoser.2018.02.014>.
- Martínez-López, J., Bagstad, K.J., Balbi, S., Magrath, A., Voigt, B., Athanasiadis, I., Pascual, M., Willcock, S., Villa, F., 2018. Towards globally customizable ecosystem service models. *Sci. Total Environ.* 650, 2325–2336. <https://doi.org/10.1016/j.scitotenv.2018.09.371>.
- Milà i Canals, L., Bauer, C., Depetele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., 2007. Key elements in a framework for land use impact assessment within LCA. *Int. J. Life Cycle Assess.* 12, 5–15. <https://doi.org/10.1065/lca2006.05.250>.

- Ministère de la mobilité et des travaux publics du Grand-Duché de Luxembourg, 2019. Occupation biophysique du sol 2007 - Portail Open Data [WWW Document]. URL: <https://data.public.lu/fr/datasets/occupation-biophysique-du-sol-2007-1/>, Accessed date: 4 February 2019.
- Muetzelfeldt, R., Massheder, J., 2003. The Simile visual modelling environment. *Eur. J. Agron.* 18, 345–358. [https://doi.org/10.1016/S1161-0301\(02\)00112-0](https://doi.org/10.1016/S1161-0301(02)00112-0).
- Müller-Wenk, R., Brandão, M., 2010. Climatic impact of land use in LCA-carbon transfers between vegetation/soil and air. *Int. J. Life Cycle Assess.* 15, 172–182. <https://doi.org/10.1007/s11367-009-0144-y>.
- Oreskes, N., Shrader-Frechette, K., Benitez, K., 1994. Verification, validation, and confirmation of numerical models in the earth sciences. *Science (80-)* 263, 641–646. <https://doi.org/10.1126/science.263.5147.641>.
- Othoniel, B., Rugani, B., Heijungs, R., Benetto, E., Withagen, C., 2016. Assessment of life cycle impacts on ecosystem services: promise, problems, and prospects. *Environ. Sci. Technol.* 50, 1077–1092. <https://doi.org/10.1021/acs.est.5b03706>.
- Pascual, U., Balvanera, P., Díaz, S., Pataki, G., Roth, E., Stenseke, M., Watson, R.T., Başak Dessane, E., Islar, M., Kelemen, E., Maris, V., Quaas, M., Subramanian, S.M., Wittmer, H., Adlan, A., Ahn, S.E., Al-Hafedh, Y.S., Amankwah, E., Asah, S.T., Berry, P., Bilgin, A., Breslow, S.J., Bullock, C., Cáceres, D., Daly-Hassen, H., Figueroa, E., Golden, C.D., Gómez-Baggethun, E., González-Jiménez, D., Houdet, J., Keune, H., Kumar, R., Ma, K., May, P.H., Mead, A., O'Farrell, P., Pandit, R., Pengue, W., Pichis-Madruga, R., Popa, F., Preston, S., Pacheco-Balanza, D., Saarikoski, H., Strassburg, B.B., van den Belt, M., Verma, M., Wickson, F., Yagi, N., 2017. Valuing nature's contributions to people: the IPBES approach. *Curr. Opin. Environ. Sustain.* 26–27, 7–16. <https://doi.org/10.1016/j.cosust.2016.12.006>.
- Pavan, A.L.R., Ometto, A.R., 2018. Ecosystem Services in Life Cycle Assessment: a novel conceptual framework for soil. *Sci. Total Environ.* 643, 1337–1347. <https://doi.org/10.1016/j.scitotenv.2018.06.191>.
- Pennington, D.W., Potting, J., Finnveden, G., Lindeijer, E., Joliet, O., Rydberg, T., Rebitzer, G., 2004. Life cycle assessment Part 2: current impact assessment practice. *Environ. Int.* 30, 721–739. <https://doi.org/10.1016/j.envint.2003.12.009>.
- Pfister, S., Koehler, A., Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. *Environ. Sci. Technol.* 43, 4098–4104. <https://doi.org/10.1021/es802423e>.
- Pfister, S., Motoshita, M., Ridoutt, B.G., 2014. Progress toward an LCA impact assessment model linking land use and malnutrition-related DALYs. Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCA Food 2014), San Francisco, California, USA, 8–10 October, 2014, pp. 1007–1015.
- Pietsch, S.A., Hasenauer, H., Thornton, P.E., 2005. BGC-model parameters for tree species growing in central European forests. *For. Ecol. Manag.* 211, 264–295. <https://doi.org/10.1016/j.foreco.2005.02.046>.
- Polasky, S., Tallis, H., Reyers, B., 2015. Setting the bar: standards for ecosystem services. *Proc. Natl. Acad. Sci.* 112, 7356–7361. <https://doi.org/10.1073/pnas.1406490112>.
- Pontius, R.G., Millones, M., 2011. Death to kappa: birth of quantity disagreement and allocation disagreement for accuracy assessment. *Int. J. Remote Sens.* 32, 4407–4429. <https://doi.org/10.1080/01431161.2011.552923>.
- Pontius, R.G., Parmentier, B., 2014. Recommendations for using the relative operating characteristic (ROC). *Landsc. Ecol.* 29, 367–382. <https://doi.org/10.1007/s10980-013-9984-8>.
- Potschin-Young, M., Haines-Young, R., Görg, C., Heink, U., Jax, K., Schleyer, C., 2018. Understanding the role of conceptual frameworks: Reading the ecosystem service cascade. *Ecosyst. Serv.* 29, 428–440. <https://doi.org/10.1016/j.ecoser.2017.05.015>.
- Pufal, G., Steffan-Dewenter, I., Klein, A.M., 2017. Crop pollination services at the landscape scale. *Curr. Opin. Insect Sci.* 21, 91–97. <https://doi.org/10.1016/j.cois.2017.05.021>.
- R Core Team, 2018. R: A Language and Environment for Statistical Computing.
- Rege, S., Arenz, M., Marvuglia, A., Vázquez-Rowe, I., Benetto, E., Igos, E., Koster, D., 2016. Quantification of agricultural land use changes in consequential life cycle assessment using mathematical programming models following a partial equilibrium approach. *J. Environ. Informatics* 26, 121–139.
- Ricketts, T.H., Regetz, J., Steffan-Dewenter, I., Cunningham, S.A., Kremen, C., Bogdansk, A., Gemmill-Herren, B., Greenleaf, S.S., Klein, A.M., Mayfield, M.M., Morandin, L.A., Ochieng', A., Viana, B.F., 2008. Landscape effects on crop pollination services: are there general patterns? *Ecol. Lett.* 11, 499–515. <https://doi.org/10.1111/j.1461-0248.2008.01157.x>.
- 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 level. *Int. J. Life Cycle Assess.* 18, 1253–1264. <https://doi.org/10.1007/s11367-013-0577-1>.
- Saarikoski, H., Mustajoki, J., Barton, D.N., Geneletti, D., Langemeyer, J., Gomez-Baggethun, E., Marttunen, M., Antunes, P., Keune, H., Santos, R., 2016. Multi-criteria decision analysis and cost-benefit analysis: comparing alternative frameworks for integrated valuation of ecosystem services. *Ecosyst. Serv.* 22, 238–249. <https://doi.org/10.1016/j.ecoser.2016.10.014>.
- Schaubroeck, T., 2017. A need for equal consideration of ecosystem disservices and services when valuing nature; countering arguments against disservices. *Ecosyst. Serv.* 26, 95–97. <https://doi.org/10.1016/j.ecoser.2017.06.009>.
- Serna-Chavez, H.M., Schulp, C.J.E., Van Bodegom, P.M., Bouten, W., Verburg, P.H., Davidson, M.D., 2014. A quantitative framework for assessing spatial flows of ecosystem services. *Ecol. Indic.* 39, 24–33. <https://doi.org/10.1016/j.ecolind.2013.11.024>.
- STATEC, 2019. Statistics portal of the Grand Duchy of Luxembourg. <https://statistiques.public.lu/en/index.html>.
- Stoldt Associés, 2018. Luxembourg 2030 - 3ème plan national pour un développement durable (projet). <https://environnement.public.lu/fr/actualites/2018/septembre2018/pnnd.html>.
- Taelman, S.E., Schaubroeck, T., De Meester, S., Boone, L., Dewulf, J., 2016. Accounting for land use in life cycle assessment: the value of NPP as a proxy indicator to assess land use impacts on ecosystems. *Sci. Total Environ.* 550, 143–156. <https://doi.org/10.1016/j.scitotenv.2016.01.055>.
- Teixeira, R.F.M., 2014. Integrating biodiversity and ecosystem services in life cycle assessment: methodological proposals for new challenges. *Chem. Eng. Trans.* 42, 127–132. <https://doi.org/10.3303/CET1442022>.
- The European Soil Database, 2001. Soil Geographical Database of Eurasia (SGDBE), Version 4 Beta. [https://esdac.jrc.ec.europa.eu/ESDB\\_Archive/ESDBv2/fr\\_intro.htm](https://esdac.jrc.ec.europa.eu/ESDB_Archive/ESDBv2/fr_intro.htm).
- Thornton, P.E., Running, S.W., Hunt Jr., E.R., 2005. Biome-BGC: Terrestrial Ecosystem Process Model, Version 4.1.1. ORNL Distributed Active Archive Center <https://doi.org/10.3334/ORNLDAAC/805>.
- Turner, K.G., Anderson, S., Gonzales-Chang, M., Costanza, R., Courville, S., Dalgaard, T., Dominati, E., Kubiszewski, I., Ogilvy, S., Porfirio, L., Ratna, N., Sandhu, H., Sutton, P.C., Svenning, J.-C.C., Turner, G.M., Varennes, Y.-D.D., Voinov, A., Wratten, S., Gonzalez-Chang, M., Costanza, R., Courville, S., Dalgaard, T., Dominati, E., Kubiszewski, I., Ogilvy, S., Porfirio, L., Ratna, N., Sandhu, H., Sutton, P.C., Svenning, J.-C.C., Turner, G.M., Varennes, Y.-D.D., Voinov, A., Wratten, S., Gonzales-Chang, M., Costanza, R., Courville, S., Dalgaard, T., Dominati, E., Kubiszewski, I., Ogilvy, S., Porfirio, L., Ratna, N., Sandhu, H., Sutton, P.C., Svenning, J.-C.C., Turner, G.M., Varennes, Y.-D.D., Voinov, A., Wratten, S., 2015. A review of methods, data, and models to assess changes in the value of ecosystem services from land degradation and restoration. *Ecol. Model.* 319, 190–207. <https://doi.org/10.1016/j.ecolmodel.2015.07.017>.
- Vázquez-Rowe, I., Marvuglia, A., Rege, S., Benetto, E., 2014. Applying consequential LCA to support energy policy: land use change effects of bioenergy production. *Sci. Total Environ.* 472, 78–89. <https://doi.org/10.1016/j.scitotenv.2013.10.097>.
- Verburg, P.H., Overmars, K.P., 2007. Dynamic simulation of land-use change trajectories with the clues-S model. In: Koomen, E., Stillwell, J., Bakema, A., Scholten, H. (Eds.), Modelling Land-Use Change Progress and Applications. Springer, Netherlands, pp. 321–335. [https://doi.org/10.1007/978-1-4020-5648-2\\_18](https://doi.org/10.1007/978-1-4020-5648-2_18).
- Verburg, P.H., Overmars, K.P., 2009. Combining top-down and bottom-up dynamics in land use modeling: exploring the future of abandoned farmlands in Europe with the dyna-CLUE model. *Landsc. Ecol.* 24, 1167–1181. <https://doi.org/10.1007/s10980-009-9355-7>.
- Verburg, P.H., Kok, K., Pontius, R.G., Veldkamp, A., 2006. Modeling Land-Use and Land-Cover Change. Land-Use and Land-Cover Change. Springer, Berlin, Heidelberg, p. 117–13. [https://doi.org/10.1007/3-540-32202-7\\_5](https://doi.org/10.1007/3-540-32202-7_5).
- van Vliet, J., Bregt, A.K., Brown, D.G., van Delden, H., Heckbert, S., Verburg, P.H., 2016. A review of current calibration and validation practices in land-change modeling. *Environ. Model. Softw.* 82, 174–182. <https://doi.org/10.1016/j.envsoft.2016.04.017>.
- von Döhrn, P., Haase, D., 2015. Ecosystem disservices research: a review of the state of the art with a focus on cities. *Ecol. Indic.* 52, 490–497. <https://doi.org/10.1016/j.ecolind.2014.12.027>.
- World Bank, 2019. Carbon pricing dashboard. <https://carbonpricingdashboard.worldbank.org/>.
- Yan, Y., 2018. Integrate carbon dynamic models in analyzing carbon sequestration impact of forest biomass harvest. *Sci. Total Environ.* 615, 581–587. <https://doi.org/10.1016/j.scitotenv.2017.09.326>.
- van Zelm, R., van der Velde, M., Balkovic, J., Čengić, M., Elshout, P.M.F., Koellner, T., Núñez, M., Obersteiner, M., Schmid, E., Huijbregts, M.A.J., 2018. Spatially explicit life cycle impact assessment for soil erosion from global crop production. *Ecosyst. Serv.* 30, 220–227. <https://doi.org/10.1016/j.ecoser.2017.08.015>.
- Zhang, Y.L., Anil, B., Bakshi, B.R., 2010. Accounting for ecosystem services in life cycle assessment part II: toward an ecologically based LCA. *Environ. Sci. Technol.* 44, 2624–2631. <https://doi.org/10.1021/es900548a>.
- Zulian, G., Maes, J., Paracchini, M., 2013. Linking land cover data and crop yields for mapping and assessment of pollination services in Europe. *Land* 2, 472–492. <https://doi.org/10.3390/land2030472>.