



## Identifying regional drivers of future land-based biodiversity footprints

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### ABSTRACT

Biodiversity footprints quantify the impacts on ecosystems caused by final consumption in a region, accounting for imports and exports. Up to now, footprint analyses have typically been applied to analyze past or present consumption patterns. Here, we quantify future land-based biodiversity footprints associated with three diverging Shared Socio-economic Pathways (SSPs), using loss in Biodiversity Intactness Index (BII) as an indicator of biodiversity loss. For each SSP, we retrieved socio-economic and land use projections to 2100 from the IMAGE-MAGNET model and calculated associated biodiversity footprints for seven aggregated world regions. We then compared these with the functional diversity component of the biosphere integrity planetary boundary. Our results indicate that the global land-based biodiversity impact stays below the boundary (tentatively set at 90% of original BII) in all scenario-year combinations. Contrastingly, the per capita boundary is transgressed in one, four and five out of the seven world regions in 2100 for SSP1 ('sustainability'), SSP2 ('middle of the road') and SSP3 ('regional rivalry'), respectively. These results indicate a strong difference in the biodiversity impact of final consumption between the regions and between SSPs. Even in the 'sustainability' scenario, the per capita biodiversity footprint of consumption in North America needs to be reduced to meet the per capita boundary. Thus, policy-making to safeguard the environment would benefit from adopting region-specific strategies: focusing on realizing agricultural efficiency gains in regions with unexploited potential, while focusing on promoting dietary changes towards less animal-based consumption in regions with limited potential for additional efficiency gains.

### 1. Introduction

Holistic approaches to address both socio-economic and environmental challenges are at the core of global integrated assessments (Doelman et al., 2019; Heck et al., 2018; Humpenöder et al., 2018; Willett et al., 2019) and target setting, such as the Sustainable Development Goals (SDGs) (UN, 2015). Scenario analyses of future states of the socio-economic system allow for evaluation of synergies and trade-offs between different social and economic development options and are useful for the assessment of different policy options (van Vuuren et al., 2012). Five Shared Socio-economic Pathways (SSPs) have been developed to provide a range of different plausible future socio-

economic development trajectories, reflecting different challenges for climate change mitigation and adaptation (Bauer et al., 2017; Riahi et al., 2017). Thus, they complement climate change assessments based on the Representative Concentration Pathways (RCPs), i.e. a range of radiative forcing trajectories used in assessments of the Intergovernmental Panel on Climate Change (IPCC) (Riahi et al., 2017). The five SSPs are characterized as: SSP1 – Taking the Green Road ('sustainability'), SSP2 – Middle of the Road, SSP3 – A Rocky Road ('regional rivalry'), SSP4 – A Road Divided ('inequality'), SSP5 – Taking the Highway ('fossil-fueled development'). Core scenario drivers are human population size, changes in gross domestic product and degrees of urbanization, which are quantified in accordance with the different SSP

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narratives (Riahi et al., 2017). These core scenario drivers are complemented by, among others, assumptions on food consumption (for instance, low animal-based consumption in SSP1; high animal-based consumption in SSP3), agricultural productivity gains (high in SSP1; low in SSP3) and international trade (global integration in SSP1; protectionism in SSP3). Consequently, scenarios differ in, for example, projected supply and demand of agricultural commodities and associated production intensities, supply and demand of different energy carriers, land use and environmental impacts (Riahi et al., 2017). The five so-called SSP baselines would result in radiative forcing levels of 5 W/m<sup>2</sup> (SSP1), 6.5 W/m<sup>2</sup> (SSP2), 6.8 W/m<sup>2</sup> (SSP3), 6.1 W/m<sup>2</sup> (SSP4) and 8.3 W/m<sup>2</sup> (SSP5), respectively, and thus require additional measures to limit global warming to below 1.5–2 degrees (corresponding to below 2.6 W/m<sup>2</sup>) (Doelman et al., 2018; Riahi et al., 2017).

Implementing the SSPs into integrated assessment models (IAMs) has generated a broad range of future trajectories of production and consumption patterns and associated shifts in energy supply and demand, land use and greenhouse gas emissions (O'Neill et al., 2017; Riahi et al., 2017; Popp et al., 2017; Van Vuuren et al., 2017). While SSP assessments typically focus on evaluating options for reaching climate targets, some recent studies have also addressed potential future land use related biodiversity impacts (for example, loss in species richness) of the SSPs (Hof et al., 2018; Powers and Jetz, 2019; Schipper et al., 2020). However, none have considered consumption-based accounting of these impacts. Consumption-based accounting links final demand (i.e. demand for goods and services by consumers and other agents in an economy) to biodiversity impacts via production and trade, using environmentally-extended multi-regional input–output (MRIO) analysis (Miller and Blair, 2009). So far, biodiversity footprint studies have focused primarily on past and current consumption patterns (Chaudhary et al., 2016; Chaudhary and Kastner, 2016; Chaudhary and Brooks, 2017; Lenzen et al., 2012; Marquardt et al., 2019; Marques et al., 2019; Moran et al., 2016; Moran and Kanemoto, 2017; Verones et al., 2017; Wilting et al., 2017). However, assessing footprints associated with socio-economic projections has the potential to provide additional insights regarding future drivers of biodiversity impacts, which in turn can inform policy-making that aims to safeguard biodiversity.

Here, we combined the fields of biodiversity footprinting and scenario-based integrated assessment modeling to quantify future land-based biodiversity footprints of three diverging SSPs, using loss in Biodiversity Intactness Index (BII) as an indicator of biodiversity loss (Newbold et al., 2016). We calculated land-based biodiversity footprints by linking BII loss to socio-economic trends and land use from the IMAGE-MAGNET model for SSP1 ('sustainability'), SSP2 ('middle of the road') and SSP3 ('regional rivalry') baselines up to the year 2100. To put the biodiversity footprint projections into the context of global biophysical limitations, we evaluated the footprints against the biosphere integrity planetary boundary, i.e. a boundary defining the safe operating space for humanity to avoid irreversible human-induced environmental change on a global scale. (Steffen et al., 2015). Our analysis provides insights about the socio-economic scenario drivers of biodiversity impacts and identifies opportunities for reducing those impacts over time in the context of a global environmental boundary.

## 2. Methods

### 2.1. Biodiversity footprints

Adding a temporal component to the methodology of Marquardt et al. (2019), we used a temporally resolved multi-regional input–output (MRIO) model to calculate the biodiversity footprints  $B$  for each of the 26 regions (Supplementary information, Table A1),  $j$ , included in the IMAGE-MAGNET model for each year  $t$  using:

$$B^{j,s,t} = mD^{s,t}(I - A^{s,t})^{-1}y^{j,s,t} + md^{j,s,t} \quad (1)$$

Land use type-specific biodiversity loss factors (in this case Biodiversity Intactness Index (BII) loss) are captured in vector  $m$  and land use intensities per region and sector for scenario  $s$  and year  $t$  in matrix  $D$ . The technology matrix  $A$  contains sector-specific information on the use of imported and domestic intermediate inputs for scenario  $s$  and year  $t$  and is the key component of the Leontief inverse,  $(I-A)^{-1}$  (Leontief, 1936). The vector  $y$  captures region  $j$ 's final demand for imported and domestic goods and services. Given the absence of appropriate data for linking urban areas (such as housing, offices etc.) to specific sectors' intermediate input demand, biodiversity impacts of urban land use were directly linked to final consumption. This link was established by adding the product  $m^*d$ ; with the vectors  $m$  and  $d$  capturing the land use type-specific biodiversity loss factors and the direct land use linked to consumption in region  $j$  for scenario  $s$  and year  $t$ , respectively.

To calculate biodiversity footprints for different years and scenarios, we used projected land use and economic data generated by the IMAGE-MAGNET modeling framework (which is based on the GTAP database) (Doelman et al., 2018; Stehfest et al., 2014). The methodology of Peters et al. (2011) was used to derive the Leontief inverse and the vector of final demand in region  $j$  for scenario  $s$  and year  $t$  from IMAGE-MAGNET model results. For our biodiversity footprint calculations, we used land use areas at 5 and 30 arc-min resolution generated by the IMAGE-LandManagement model and the aforementioned economic data at a resolution of 26 regions and 30 sectors comprising different types of industry generated by the MAGNET general equilibrium model (Woltjer et al., 2014). These data were transformed in two steps. First, the land use data for cropland, pasture, forest and urban areas were harmonized with the land use intensity classification of the biodiversity indicator (Supplementary information, Table A1). Second, land use areas were allocated to the relevant economic sectors and regions (see Supplementary information, Section A3 for a detailed description). For easier evaluation, we aggregated the results to seven world regions and seven consumption categories (Supplementary information, Table A1 and Table A3)

We calculated biodiversity footprints as loss in terrestrial biodiversity intactness based on the Biodiversity Intactness Index (BII) indicator by Newbold et al. (2016). This indicator reflects loss of abundance, correcting for differences in compositional similarity (i.e. accounting for differences in species' occurrence in the different land use types) between the natural reference situation and various land use types (primary vegetation, secondary vegetation, cropland, pasture and urban) and intensities (minimal, light, intense) (Newbold et al., 2016; see also Supplementary information, Table A2 for an overview of the applied BII loss factors). Following Newbold et al. (2016), resulting BII loss factors provide a time-independent characterization of the biodiversity intactness of each land use type relative to the undisturbed natural reference situation. Thus, linking projected land use areas to the BII loss factors allows for quantification of future biodiversity intactness (see, for example, Leclère et al. 2020) and hence changes in land-related biodiversity footprints.

### 2.2. IMAGE-MAGNET model projections

The IMAGE-MAGNET model framework connects various models (ranging from the socio-economic general equilibrium model MAGNET (Woltjer et al., 2014) to the vegetation model LPJmL (Müller et al., 2016)) to project long-term interactions between human activities and the environment (Doelman et al., 2018; Stehfest et al., 2014). The individual models are linked by exchanging certain input and/or output data. For example, MAGNET simulates future demand for agricultural output (crops and livestock) that is passed on to the IMAGE-LandManagement model to determine how this demand is met (Doelman et al., 2018).

At the core of our MRIO framework were the results generated by the MAGNET model. Grounded in neo-classical microeconomic theory, MAGNET determines input and output prices for all sectors and

economic agents to arrive at a projected equilibrium of supply and demand (Woltjer et al., 2014). Given its sectoral focus on agricultural commodities, MAGNET provides comprehensive results of projected agricultural production, consumption and trade (Doelman et al., 2018). Core drivers of the model are assumptions on GDP, population and technological progress as defined by the scenario set-up, in our case, the narratives of the SSPs. In addition, scenario assumptions on land use, future dietary preferences and trade policies are used. Projected consumption patterns and volumes are then derived from consumption functions (Woltjer et al., 2014). These depend on available income, prices and income elasticities provided by the GTAP database (Hertel and van der Mensbrugge, 2016). In MAGNET, the elasticities are adjusted so that the share of future spending on food, as a proportion of total income, decreases while the share of service consumption increases as a result of growing income (Engel's law – Engel, 1857; Woltjer et al., 2014). Trade mainly results from total economic growth (driven by GDP assumptions) and related income growth. The sourcing of products is determined by the economic agents' ability to substitute (determined by substitution elasticities) domestic and imported products given changes in relative prices of these products. Regional trade patterns are determined by Armington elasticities that imply heterogeneous preferences for domestic and imported products per country of origin (Armington, 1969; Hertel and van der Mensbrugge, 2016). This approach to model international trade patterns has been extensively validated (Ahhammad et al., 2015; Francois et al., 2005; van Meijl and Van Tongeren, 2004).

The second key component in our MRIO was the land use areas derived by the IMAGE-LandManagement model. To meet the MAGNET model's projected demand for crops and livestock, the IMAGE-LandManagement model simulates the required land use intensification and extensification levels and livestock system changes (Doelman et al., 2018). Location-specific intensification and extensification of cropland and pasture is determined by historical land cover, potential crop yields (based on LPJmL projections) and location-specific features (such as accessibility and terrain slope) (Doelman et al., 2018). For livestock, the IMAGE-LandManagement model distinguishes between intensive and extensive livestock management systems. Simulated changes in livestock systems account for region-specific feed composition and efficiency, animal productivity etc. and thus influence the demand for grassland (Bouwman et al., 2005). To classify cropland and pasture areas according to the intensity levels relevant for the connection with the biodiversity loss factors, we relied on grid cell specific synthetic and manure nitrogen fertilizer application rates (in MgN per km<sup>2</sup>) generated by IMAGE. We distinguished three intensity levels: low (<50 kgN per ha), medium (50–150 kgN per ha), high (>150 kgN per ha) (based on Overmars et al., 2014 and Temme and Verburg, 2011).

The IMAGE-LandManagement model distinguishes three forest management systems – clear cut, selective logging and forest plantation. Since the MAGNET model only includes one forestry sector, all forest land use areas were allocated to this sector. Thus, resulting biodiversity impacts correspond to a cumulative footprint of the three forest management systems.

### 2.3. SSPs

For the present analysis, we calculated land-based biodiversity footprints for SSP1, SSP2 and SSP3 based on IMAGE-MAGNET model results as described in Doelman et al. (2018). We selected SSP1 ('sustainability') and SSP3 ('regional rivalry') because these showed the most extreme land use changes among the five SSPs (Doelman et al., 2018). In addition, we chose SSP2 ('middle of the road') to represent business as usual with moderate socio-economic development and land use changes. Below we outline the key features of the three SSPs relevant for our analysis. For a more detailed description of the land projections and underlying assumptions, we refer to Doelman et al. (2018). For more information on general land use features of the SSP scenarios, we refer to Popp et al. (2017) and Stehfest et al. (2019).

As the SSP1 storyline assumes reduced animal-based consumption, a 30% reduction of animal-based products (compared to the endogenous outcome without a consumption-side preference shift, i.e. the SSP2 outcome) has been implemented as a preference shift in the MAGNET scenario specifications. For SSP3, a 30% increase is assumed (again, compared to the SSP2 outcome). In addition to this, SSP1 assumes a decrease in food losses of 33%, while SSP3 assumes a 33% increase.

Efficiency developments for crops and livestock in SSP2 were based on region-specific yield and livestock efficiency projections from the FAO Agricultural Outlook (Alexandratos and Bruinsma, 2012). The estimated regression between GDP and efficiency trends in SSP2 was used to project efficiency trends for SSP1 and SSP3 (based on the scenarios' assumed GDP trends). For crops, temporal trends in yields were based on autonomous technological change (e.g. based on improved technology and management) and price-driven adjustments (i.e. linked to substitution of input factors and land allocation per grid cell) (van Zeist et al., 2020). Following Doelman et al. (2018), 50% of yield improvements were assumed to be autonomous and 50% were assumed to be price-driven.

The degree of protectionism of the different scenario narratives is reflected by the implemented trade barriers (tariffs, quotas etc.). While the SSP2 implementation assumes coherence with current trade agreements and barriers, food self-sufficiency concerns of the SSP3 assume a 10% increase in import taxes for agri-food products compared with the 'middle of the road' (SSP2) situation (Doelman et al., 2018). For SSP1, it was assumed that all tariffs were removed, i.e. reflecting a highly globalized world.

### 2.4. Performance in relation to the biosphere integrity planetary boundary

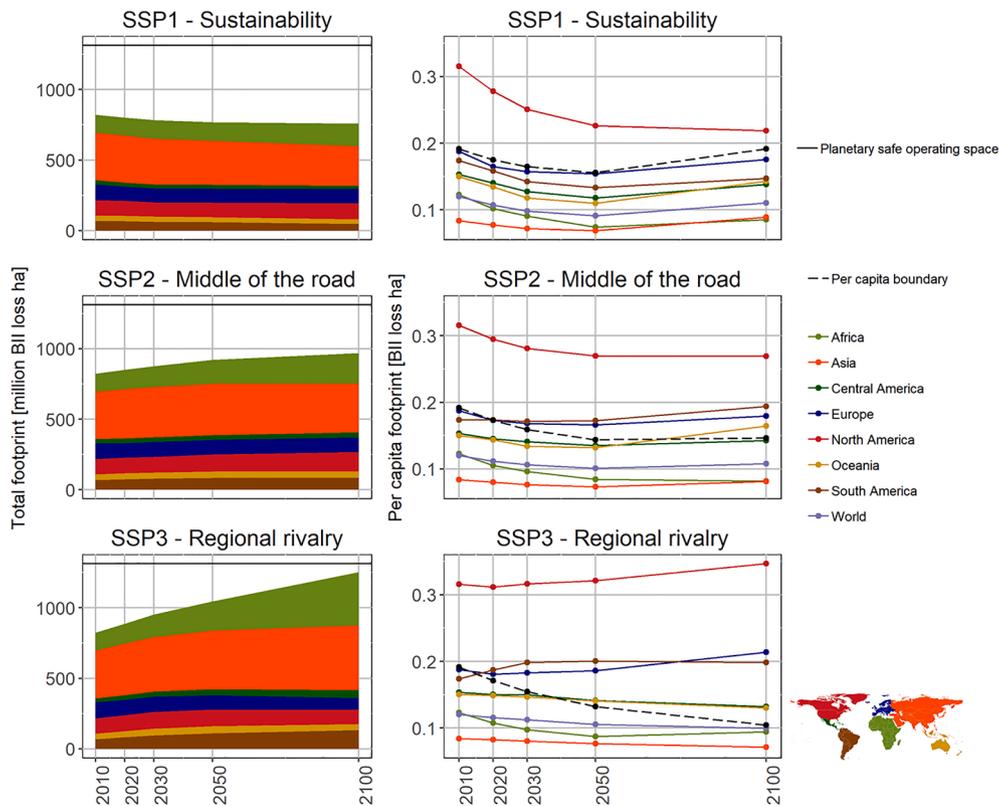
To put our assessment of land-based biodiversity footprints in the context of global environmental constraints, we compared the footprints to the functional biodiversity component of the biosphere integrity planetary boundary. We did this by applying a per capita sharing principle, dividing the planetary safe operating space by the global human population. The safe operating space (tentatively set at 90% of original BII; Steffen et al. 2015) corresponds to a BII loss allowance of 10%, equivalent to 1.31 billion hectares of water and ice-free natural land area (Steffen et al., 2015). Scenario- and year-specific population numbers resulted in a changing per capita share of the safe operating space and thus changing per capita boundaries (dashed line in Fig. 1), against which we evaluated the regions' per capita biodiversity footprints.

## 3. Results

### 3.1. Land-based biodiversity footprints over time

In SSP1, both the global total and per capita biodiversity footprints declined from 2010 to 2100 (Fig. 1; Fig. A8 in Supplementary information). For SSP2 and SSP3, the global total footprint increased by around 18% and 52%, respectively, while the global per capita footprints declined (Supplementary information, Fig. A8). Lower population contributed to lower total and per capita footprints for SSP1. Contrastingly, stronger population increases, compared to the increase in total footprints, contributed to the net declines in per capita footprints for SSP2 and SSP3 (Supplementary information, Fig. A8).

While four regions (Europe, Oceania, Central America, South America) achieved a net decline in both (total and per capita) footprints for SSP1, only Europe showed a net decline in both (total and per capita) footprints for SSP2 and none of the regions for SSP3. None of the regions showed a net increase in both (total and per capita) footprints for SSP1; two for SSP2 (Oceania and South America) and one for SSP3 (South America). For both Oceania and South America, population decreased in SSP1 (Oceania: –12%; South America: –19%) while it increased in SSP2 and SSP3, respectively (Oceania: 6% and 24%; South America: 10% and



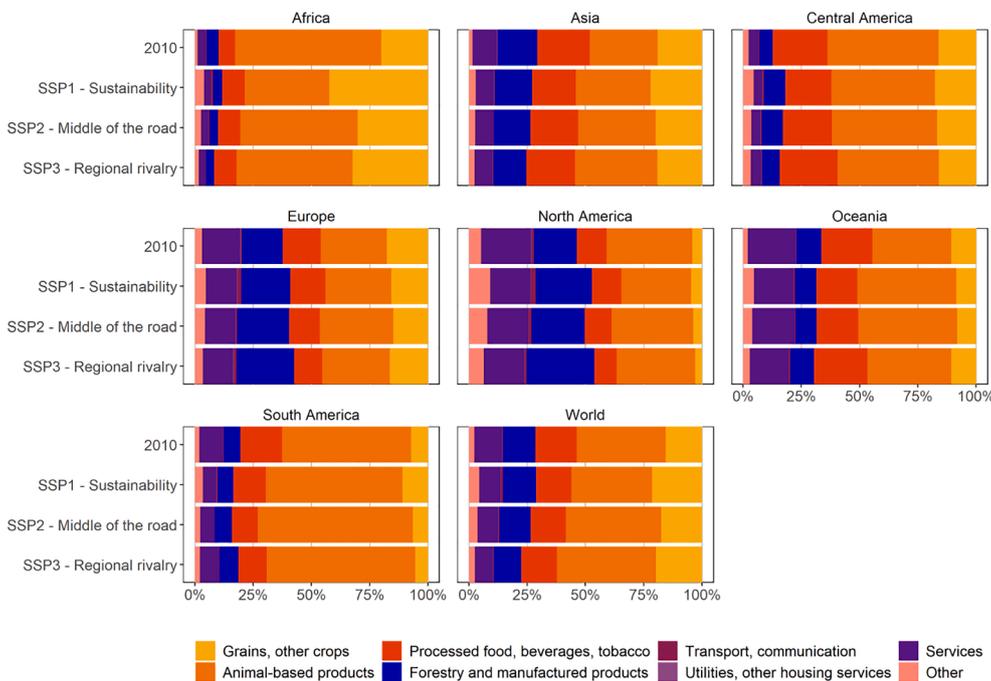
**Fig. 1.** Total and per capita land-based biodiversity footprints over time (2010, 2020, 2030, 2050 and 2100) of the SSP1 ('sustainability'), SSP2 ('middle of the road') and SSP3 ('regional rivalry') baselines. Biodiversity footprints represent losses in biodiversity intactness as quantified by the Biodiversity Intactness Index (BII). *Note:* The per capita boundary changes over time and per scenario due to changes in human population size.

70%) (Supplementary information, Section A6).

Africa's per capita footprints declined for all three scenarios, yet its total footprints showed a net increase, particularly for SSP3, due to a strong population increase (Supplementary information, Fig. A1). Contrastingly, some regions showed a net decrease in total footprints while having net increasing per capita footprints linked to a decrease in

population (Asia, SSP1; Europe and North America, SSP3).

Benchmarking regions' per capita footprints against the scenario- and year-specific per capita planetary boundary showed that the world on average stayed below the boundary in all scenarios and years. North America was the only region to consistently exceed the per capita boundary, while Africa and Asia stayed below the boundary in all three



**Fig. 2.** Contribution of consumption categories to regions' land-based biodiversity footprints for 2010 and 2100 for the SSP1 ('sustainability'), SSP2 ('middle of the road') and SSP3 ('regional rivalry') baselines. Biodiversity footprints represent losses in biodiversity intactness as quantified by the Biodiversity Intactness Index (BII). *Note:* The breakdown per consumption category omits biodiversity losses directly allocated to the final demand (i.e. losses that are not connected to the intermediate demand matrix, i.e. the demand of a specific sector, of the MRIO framework and that are not traded). For completeness, these losses have been included in Fig. 2 and are labeled as 'Other'.

scenarios and all years. However, scenario-specific trends were observed: for SSP1 declining per capita footprints brought North America closer to the boundary; for SSP3 increasing per capita footprints moved it further above the boundary. Four regions (North America, Europe, South America, Oceania) transgressed the boundary in 2100 in SSP2 and five regions in SSP3 (the aforementioned regions and Central America). The observation that the world, on average, stays below the boundary in SSP3 indicates that the relatively low consumption in the most populous regions (Africa and Asia – with 83% of global population in 2100) compensates for the consumption-based biodiversity losses of the five transgressing regions.

### 3.2. Footprint changes per consumption category

Consumption of agricultural produce ('grains, other crops' and 'animal-based products') was the dominant contributor to regions' footprints in 2010, ranging from 41% for North America to 83% for Africa (Fig. 2; Supplementary information, Table A5). For Central America and Oceania, the consumption of processed foods accounted for a considerable share as well (22–24%), while the consumption of 'forest and manufactured products' and 'services' was also important for Europe and North America (34–40%).

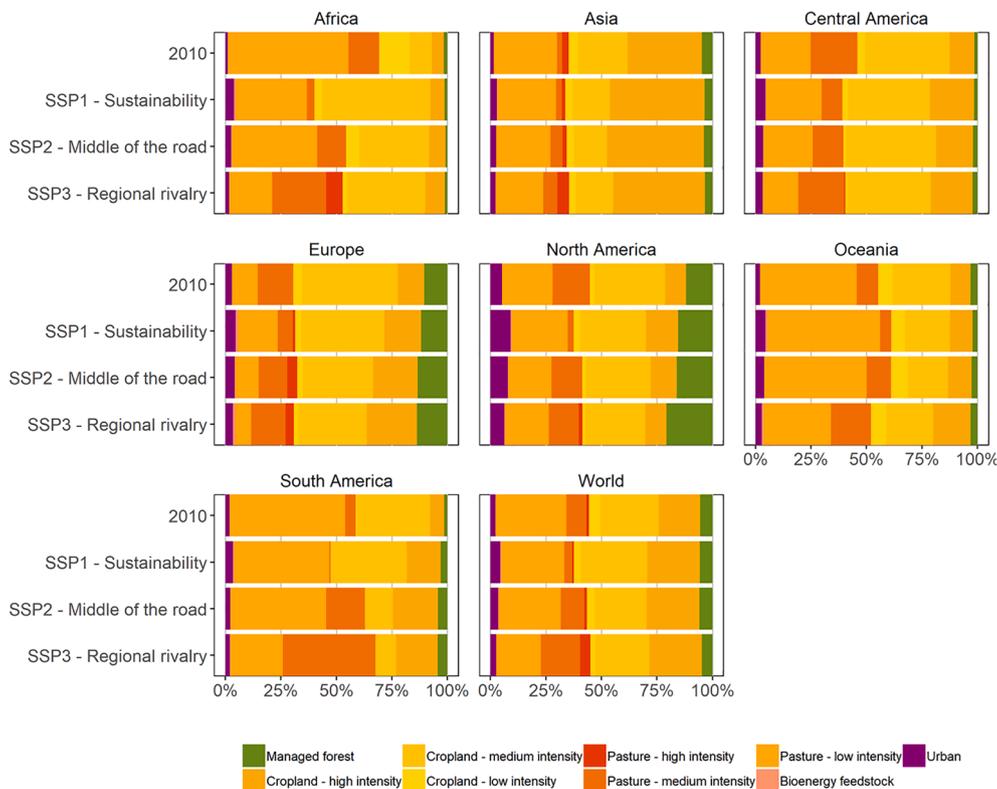
Across the three SSPs, the region-specific consumption categories that were dominant in 2010 remained dominant in 2100. Generally, the decline in footprints in the sustainability scenario was associated with lower footprints of agricultural produce (particularly 'animal-based products') (Fig. 2; Supplementary information, Table A5). However, the contribution of consumption of agricultural produce (particularly 'animal-based products') increased in Oceania (from 45% in 2010 to 51% in 2100) and Asia (from 48% in 2010 to 54% in 2100). For Africa, the contribution of agricultural produce remained fairly constant in SSP1 (from 83% in 2010 to 79% in 2100), yet with compensatory trends for 'grains, other crops' (increasing from 20% to 42%) and 'animal-based

products' (decreasing from 63% to 36%). For most of the regions, SSP2 resembled SSP1 in terms of shifts in the contribution of agricultural produce to regions' footprints. For South America, however, we found increases in footprints primarily driven by higher 'animal-based products' consumption (increasing from 55% in 2010 to 67% in 2100 for SSP2). This increase was even stronger in SSP3 (Fig. 2, Supplementary information, Table A5).

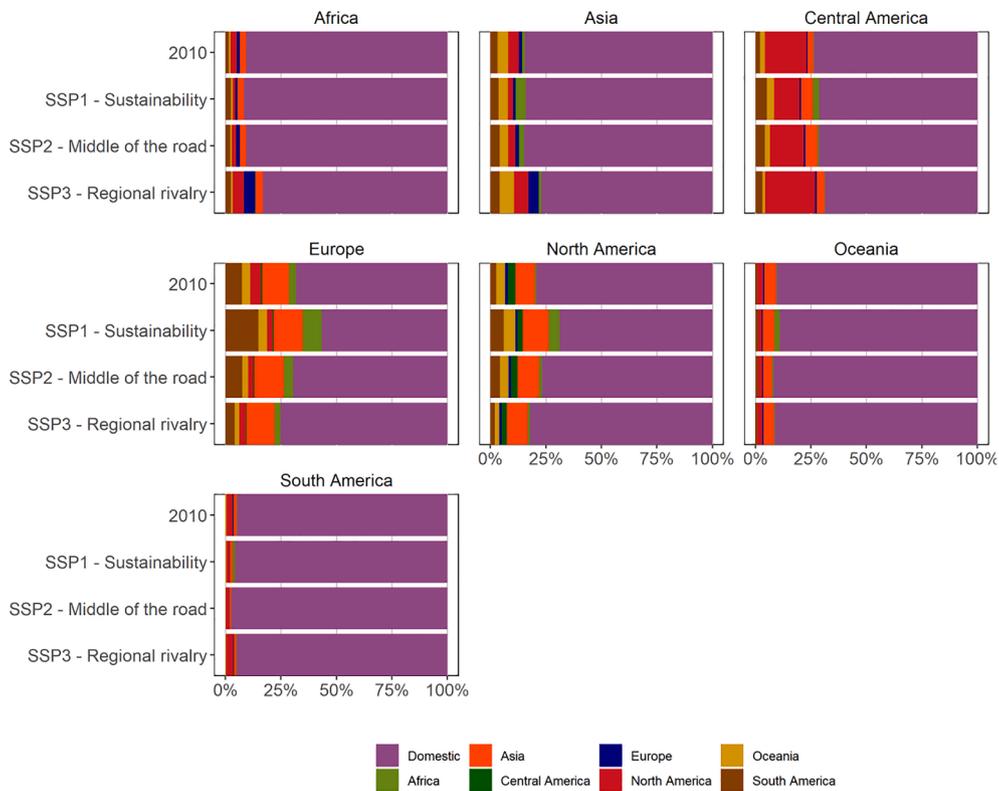
### 3.3. Contributions of different land-use types

Agricultural land use determined regions' biodiversity footprints in 2010 with contributions ranging from 83% in North America to 97% in Africa and South America (Fig. 3; Supplementary information, Section A8). Cropland-related consumption contributed on average around 50% (ranging from 29% for Africa to 60% for Asia) and pasture-related consumption on average around 42% (ranging from 28% Europe to 68% Africa). The contributions of consumption linked to different land use intensities varied per region: low-intensity agricultural land use dominated for Africa, while medium- to high-intensity was most important in the other world regions.

While the shares of cropland vs pasture remained largely the same between 2010 and 2100, there were shifts in the contributions of different land use types. For consumption in Africa, for example, we found a larger contribution of medium-intensity cropland for SSP1, primarily linked to domestic cropland intensification (given the low shares of import-related biodiversity loss (Fig. 4; Supplementary information, Section A9), a direct link can be drawn to Africa's land footprints (Supplementary information, Section A8). The contribution of consumption related to low-intensity pasture declined in SSP1, linked to a generally declining relevance of animal-based consumption (Fig. 2). Contrastingly, the increase in the contribution of animal-based consumption in SSP2 and SSP3 was linked to higher contributions of medium- and, for SSP3, also high-intensity pasture. For other regions, such



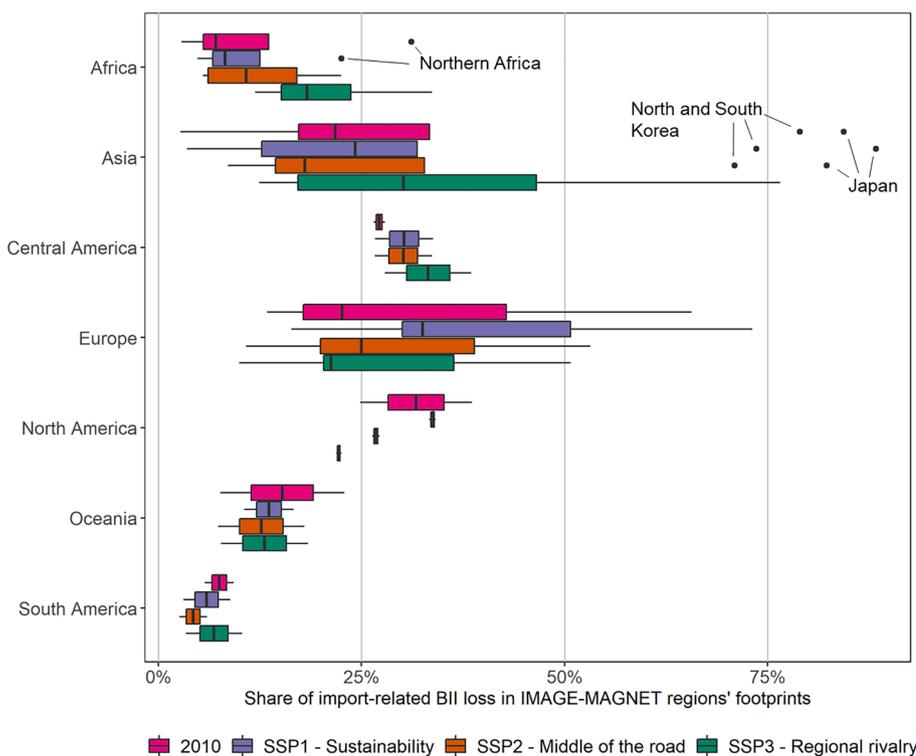
**Fig. 3.** Contribution of land use types to regions' land-based biodiversity footprints for 2010 and 2100 for the SSP1 ('sustainability'), SSP2 ('middle of the road') and SSP3 ('regional rivalry') baselines. Biodiversity footprints represent losses in biodiversity intactness as quantified by the Biodiversity Intactness Index (BII). Note: Bioenergy feedstock related consumption contributes on average around less than 1% in 2010 (compare Supplementary information, Section A8).



**Fig. 4.** Import-related biodiversity loss per origin in regions' land-based biodiversity footprints for 2010 and 2100 for the SSP1 ('sustainability'), SSP2 ('middle of the road') and SSP3 ('regional rivalry') baselines. Biodiversity footprints represent losses in biodiversity intactness as quantified by the Biodiversity Intactness Index (BII).

as Europe, we found a decline of low- to medium-intensity and an increase in high-intensity cropland-related biodiversity footprints in SSP1, reflecting cropland intensification. Lower animal-based consumption also resulted in a lower contribution of pasture-related consumption in

biodiversity footprints for Europe for the SSP1 scenario. Similar results were observed for Europe's SSP2 biodiversity footprints, while the SSP3 biodiversity footprints were driven by an intensification of pasture and thus a corresponding decline of low-intensity pasture.



**Fig. 5.** Spread of import-related biodiversity loss shares in the 26 disaggregated IMAGE-MAGNET regions' land-based biodiversity footprints (grouped per world region) for 2010 and 2100 for the SSP1 ('sustainability'), SSP2 ('middle of the road') and SSP3 ('regional rivalry') baselines. Biodiversity footprints represent losses in biodiversity intactness as quantified by the Biodiversity Intactness Index (BII). *Note:* Africa encompasses four regions; Asia nine; Central America two; Europe five; North America two; Oceania two; South America two (Supplementary information, Section A1).

### 3.4. Footprints over time and the role of import-related biodiversity losses

For all seven world regions, 2010 footprints were primarily linked to biodiversity losses of domestic origin, with trade contributions between 5% for South America to 32% for Europe (Fig. 4; Supplementary information, Table A7a). Looking at the disaggregated 26 regions in the IMAGE-MAGNET model showed a considerable spread of the relevance of import-related biodiversity loss with import-related shares up to 84% for Japan (Fig. 5; Supplementary information, Table A7b).

World regions for which import-related biodiversity loss was of limited relevance in 2010, also showed limited import-related biodiversity loss for SSP1 in 2100. Contrastingly, import-related biodiversity losses increased for Europe (from 32% to 43%), Central America (from 26% to 29%), and North America (from 21% to 31%) in the SSP1 scenario. The higher share of import-related biodiversity loss in the SSP1 footprints reflect an increased specialization of regions' production and an increasing relevance of international trade due to lower trade barriers. However, as indicated by the comparatively lower total and per capita footprints for these regions (compared to the SSP2 footprints), this finding does not necessarily coincide with higher footprints. Contrastingly, the contribution of import-related biodiversity loss remained stable for the SSP2 footprints and showed diverging trends for the SSP3 footprints. For SSP3, the relevance of imports increased for regions with initially low contributions of import-related biodiversity loss (Africa: from 9% to 17%; Asia from 16% to 23%) and showed opposite trends for regions with initially high contributions of import-related biodiversity loss (Europe: declining from 32% to 25%).

## 4. Discussion

### 4.1. Biodiversity footprints over time

Four out of seven regions achieved net total and per capita biodiversity footprint declines for the SSP1 'sustainability' scenario in 2100. Only Europe achieved net declines for both total and per capita footprints for the SSP2 'middle of the road' scenario and no region saw declines for both footprint types for the SSP3 'regional rivalry' scenario. Except for SSP1, regional footprint trends did not show converging patterns to lower footprint levels among regions (compare Fig. 1). Thus, region-specific drivers determine differences in results. Animal-based consumption (low in SSP1; high in SSP3) is a key driver for differences in the European total and per capita footprints. Changes in population growth (declining in SSP1; increasing in SSP3) and the ability to realize agricultural efficiency gains (high in SSP1; low in SSP3) are key scenario drivers of total and per capita footprint results for Africa. Furthermore, our results for the SSP1 scenario (particularly in comparison to SSP3 results) support findings that lower population growth rates (Billen et al., 2015; Doelman et al., 2018) and changes in consumption patterns (Willett et al., 2019) can alleviate resource constraints and facilitate more sustainable and less biodiversity impactful development. These drivers have also been identified as key to explain historic changes in biodiversity footprints (Marques et al., 2019) and status quo footprint assessments (Marquardt et al., 2019; Wilting et al., 2017). Finally, looking at the results of an isolated effect of scenario-year specific technical change matrix (A matrix in Eq. (1)) developments, shows reductions for SSP1 footprints over time and increases in most regions for the SSP3 footprints (Supplementary information, Section A10). This aligns with the narratives of the scenario implementation: rapid technical change improvements and trade liberalization for SSP1; stagnating technical change improvements and an increase of trade barriers for SSP3 (Doelman et al., 2018).

Improvements in agricultural efficiency (in contrast to intensification as a means to increase agricultural production) can be beneficial for biodiversity by lowering land demand (Doelman et al., 2018; Popp et al., 2017). However, the BII indicator assigns a higher threat to biodiversity arising from high-intensity agricultural areas (compare the

higher BII loss factors for medium and high intensity cropland and pasture in Table A2 of the Supplementary information and the higher shares of high-intensity areas in the footprints (Fig. 3)) which might erode the benefits of an overall lower land demand achieved through intensification. This emphasizes the importance of minimizing absolute land demand alongside shifts in other drivers (e.g. adopting less biodiversity impactful dietary patterns such as a move to plant-based diets and supply chain improvements such as a reduction in food losses and waste).

Our footprint calculations only account for land-based biodiversity impacts and neglect other possible drivers of biodiversity loss, including climate change as one of the main emerging threats to global biodiversity (IPBES, 2019; Maxwell et al., 2016; Schipper et al., 2020). Indeed, climate change not only affects biodiversity directly but also indirectly via changing agricultural yields and the associated implications for land demand (Asseng et al., 2013; Müller et al., 2015; Nelson et al., 2014; Rosenzweig et al., 2014). Thus, future scenario-based footprint studies can be improved by including additional drivers of biodiversity loss.

In addition, we recommend using additional indicators of biodiversity loss to better capture the multi-dimensional nature of biodiversity (Díaz et al., 2020; Marquardt et al., 2019; Schipper et al., 2016; Vačkář et al., 2012). The BII represents local assemblage-level intactness, i.e. covering only one aspect of biodiversity, and neglecting, for example, potential spatial variability in biodiversity impacts (Faith et al., 2008; Newbold et al., 2016). Consequently, biodiversity assessments can be improved by including additional and complementary indicators alongside the BII, covering, for example, species richness or spatial turnover of species (Marquardt et al., 2019; Vačkář et al., 2012). The relevance of covering multiple biodiversity dimensions is reflected in the Planetary Boundaries framework as the biosphere integrity planetary boundary includes both local biodiversity intactness, as an indicator of functional diversity, and global extinction rate, as an indicator of genetic diversity (Steffen et al., 2015).

### 4.2. Planetary boundary context

Benchmarking the per capita land-based biodiversity footprints against the per capita functional diversity component of the biosphere integrity planetary boundary showed that while one (SSP1) to five (SSP3) regions transgressed the boundary in 2100 in the different scenarios, the global footprint stayed below the planetary boundary in all scenarios and all years (Fig. 1). The findings for the global footprint contrast findings by Newbold et al. (2016) who found that global average abundance exceeded the planetary boundary in 2005 (BII of 84.6%). Newbold et al. (2016) also accounted for impacts linked to human population density and distance to roads, which may explain the differences.

Using socio-economic sharing principles to allocate the global safe operating space to different land-based activities enables us to link sub-global stakeholder action and global (or regional) environmental sustainability objectives (Downing et al., 2019; Häyhä et al., 2016; O'Neill et al., 2018). Several authors have considered various sharing principles with different underlying ethical rationales (Bjørn et al., 2020; Häyhä et al., 2016; Lucas et al., 2020; Nykvist et al., 2013; Ryberg et al., 2018), but a broader societal consensus on such choices (when and how to apply) is still missing (Clift et al., 2017).

In addition to identifying appropriate sharing principles for sub-global assessments of the planetary boundary, further work is needed to account for the genetic diversity component of the biosphere integrity boundary. Rockström et al. (2009) and Steffen et al. (2015) have identified global species extinction rates as an interim indicator. However, to our knowledge, there is a current lack of data and approaches to adequately assess biodiversity loss against the genetic diversity component of the biosphere integrity boundary. An adequate assessment of extinction rates would require knowledge of the number of extinctions and the timeframe from the initial disruption (e.g. land use

change) to the full extinction of the affected species.

## 5. Implications and outlook

Given the large and increasing human impacts on the environment, comprehensive environmental assessments should account for socio-economic dynamics (Häyhä et al., 2016). Our results for the more sustainable future scenario (SSP1) highlight that both production- and consumption-side changes can play a key role in lowering humanity's environmental footprint and thus contribute towards more sustainable production and consumption patterns (as advocated by SDG 12). For regions with high unexploited potential for production efficiency gains and dynamic population changes, such as Africa, production side measures are key for lowering environmental impacts. For example, intensification can serve as a driver for lowering land demand (and consequently biodiversity impacts) whilst increasing agricultural output and food security for a growing population (SDG 2.3; Heck et al., 2018). Future policy action should focus on overcoming barriers to closing yield gaps as well as supply chain measures for preserving these benefits through to consumption (for example, lowering harvest and post-harvest losses). Additionally, greater emphasis is needed to promote best agricultural management practices for mitigating some of the additional biodiversity impacts associated with high intensity agriculture.

For regions, such as Europe and North America, with limited potential for future production efficiency improvements yet high environmental impacts, consumption-side measures, such as continued advocacy related to dietary shifts, are needed to lower environmental impacts. As indicated by our results, particularly lower animal-based consumption is beneficial for lowering land-based biodiversity footprints. However, the ability to scale plant-based protein substitutes requires consideration of both knock-on health (Stehfest et al., 2009; van Vuuren et al., 2018), socio-economic and environmental effects and in some regions there may be opportunities for substituting farmed meat with less conventional meat sources (e.g. kangaroo, see Ratnasiri and Bandara (2017)). Such considerations should feature more strongly in the formulation of region-specific dietary options.

Finally, linking socio-economic-biodiversity assessments to targets and benchmarks that are recognized by a wider community (public and private decision-makers) and are grounded in Earth system science, such as the sustainable development goals and the planetary boundaries (Downing et al., 2019; Galaz et al., 2012; Häyhä et al., 2016; Heck et al., 2018), should help to ensure increased recognition of biodiversity concerns in decision-making and society in general. Such widespread understanding of environmental concerns is needed to motivate large-scale shifts towards more sustainable production and consumption patterns.

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## CRediT authorship contribution statement

**Sandra G. Marquardt:** Conceptualization, Methodology, Software, Formal analysis, Data curation, Writing - original draft. **Jonathan C. Doelman:** Data curation, Writing - review & editing. **Vassilis Daioglou:** Data curation, Writing - review & editing. **Andrzej Tabeau:** Data curation, Writing - review & editing. **Aafke M. Schipper:** Writing - review & editing, Formal analysis. **Sarah Sim:** Conceptualization, Writing - review & editing. **Michał Kulak:** Conceptualization, Writing - review & editing. **Zoran J.N. Steinmann:** Conceptualization, Software, Writing - review & editing. **Elke Stehfest:** Writing - review & editing. **Harry C. Wilting:** Methodology, Writing - review & editing. **Mark A.J. Huijbregts:** Conceptualization, Writing - review & editing.

## Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.gloenvcha.2021.102304>.

## References

- Ahammad, H., Heyhoe, E., Nelson, G., Sands, R., Fujimori, S., Hasegawa, T., van der Mensbrugge, D., Blanc, E., Havlik, P., Valin, H., Kyle, P., Mason-D'Croz, D., van Meijl, J.C.M., Schmitz, C., Lotze-Campen, H., von Lampe, M., Tabeau, A.A., 2015. The role of international trade under a changing climate: insights from global economic modelling. In: Elbehri, A. (Ed.), *Climate Change and Food Systems: Global Assessments and Implications for Food Security*. FAO, pp. 293–312.
- Alexandratos, N., Bruinsma, J., 2012. *World Agriculture towards 2030/2050: the 2012 Revision*. ESA Working Paper No. 12-03. FAO, Rome.
- Armington, P.S., 1969. A theory of demand of products distinguished by place of production. *IMF Econ. Rev.* 16, 159–178. <https://doi.org/10.2307/3866403>.
- Asseng, S., Ewert, F., Rosenzweig, C., Jones, J.W., Hatfield, J.L., Ruane, A.C., Boote, K.J., Thorburn, P.J., Rötter, R.P., Cammarano, D., Brisson, N., Basso, B., Martre, P., Aggarwal, P.K., Angulo, C., Bertuzzi, P., Biernath, C., Challinor, A.J., Doltra, J., Gayler, S., Goldberg, R., Grant, R., Heng, L., Hooker, J., Hunt, L.A., Ingwersen, J., Izaurralde, R.C., Kersebaum, K.C., Müller, C., Naresch Kumar, S., Nendel, C., O'Leary, G., Olesen, J.E., Osborne, T.M., Palosuo, T., Priesack, E., Ripoche, D., Semenov, M.A., Shcherbak, I., Steduto, P., Stöckle, C., Stratonovitch, P., Streck, T., Supit, I., Tao, F., Travasso, M., Waha, K., Wallach, D., White, J.W., Williams, J.R., Wolf, J., 2013. Uncertainty in simulating wheat yields under climate change. *Nat. Clim. Change* 3 (9), 827–832. <https://doi.org/10.1038/nclimate1916>.
- Bauer, N., Calvin, K., Emmerling, J., Fricko, O., Fujimori, S., Hilaire, J., Eom, J., Krey, V., Kriegler, E., Mouratiadou, I., Sytze de Boer, H., van den Berg, M., Carrara, S., Daioglou, V., Drouet, L., Edmonds, J.E., Gernaat, D., Havlik, P., Johnson, N., Klein, D., Kyle, P., Marangoni, G., Masui, T., Pietzcker, R.C., Strubegger, M., Wise, M., Riahi, K., van Vuuren, D.P., 2017. Shared socio-economic pathways of the energy sector – quantifying the narratives. *Global Environ. Change* 42, 316–330. <https://doi.org/10.1016/j.gloenvcha.2016.07.006>.
- Billen, G., Lassaletta, L., Garnier, J., 2015. A vast range of opportunities for feeding the world in 2050: trade-off between diet, N contamination and international trade. *Environ. Res. Lett.* 10 (2), 025001. <https://doi.org/10.1088/1748-9326/10/2/025001>.
- Björn, A., Sim, S., Boulay, A.-M., King, H., Clavreul, J., Lam, W.Y., Barbarossa, V., Bulle, C., Margni, M., 2020. A planetary boundary-based method for freshwater use in life cycle assessment: development and application to a tomato production case study. *Ecol. Ind.* 110, 105865. <https://doi.org/10.1016/j.ecolind.2019.105865>.
- Bouwman, A.F., Van der Hoek, K.W., Eickhout, B., Soenario, I., 2005. Exploring changes in world ruminant production systems. *Agric. Syst.* 84 (2), 121–153. <https://doi.org/10.1016/j.agry.2004.05.006>.
- Chaudhary, A., Brooks, T.M., 2017. National consumption and global trade impacts on biodiversity. *World Dev.* 121, 178–187. <https://doi.org/10.1016/j.worlddev.2017.10.012>.
- Chaudhary, A., Kastner, T., 2016. Land use biodiversity impacts embodied in international food trade. *Global Environ. Change* 38, 195–204. <https://doi.org/10.1016/j.gloenvcha.2016.03.013>.
- Chaudhary, A., Pfister, S., Hellweg, S., 2016. Spatially explicit analysis of biodiversity loss due to global agriculture, pasture and forest land use from a producer and consumer perspective. *Environ. Sci. Technol.* 50 (7), 3928–3936. <https://doi.org/10.1021/acs.est.5b06153>.
- Clift, R., Sim, S., King, H., Chenoweth, J., Christie, I., Clavreul, J., Mueller, C., Posthuma, L., Boulay, A.-M., Chaplin-Kramer, R., Chatterton, J., DeClerck, F., Druckman, A., France, C., Franco, A., Gerten, D., Goedkoop, M., Hauschild, M., Huijbregts, M., Koellner, T., Lambin, E., Lee, J., Mair, S., Marshall, S., McLachlan, M., Milà i Canals, L., Mitchell, C., Price, E., Rockström, J., Suckling, J., Murphy, R., 2017. The challenges of applying planetary boundaries as a basis for strategic decision-making in companies with global supply chains. *Sustainability* 9 (2), 279. <https://doi.org/10.3390/su9020279>.
- Díaz, S., Zafra-Calvo, N., Purvis, A., Verburg, P.H., Obura, D., Leadley, P., Chaplin-Kramer, R., De Meester, L., Dulloo, E., Martín-López, B., Shaw, M.R., Visconti, P., Broadgate, W., Bruford, M.W., Burgess, N.D., Cavender-Bares, J., DeClerck, F., Fernández-Palacios, J.M., Garibaldi, L.A., Hill, S.L.L., Isbell, F., Khoury, C.K., Krug, C.B., Liu, J., Maron, M., McGowan, P.J.K., Pereira, H.M., Reyes-García, V., Rocha, J., Rondinini, C., Shannon, L., Shin, Y.-J., Snelgrove, P.V.R., Spehn, E.M., Strassburg, B., Subramanian, S.M., Tewksbury, J.J., Watson, J.E.M., Zanne, A.E., 2020. Set ambitious goals for biodiversity and sustainability. *Science* 370 (6515), 411–413. <https://doi.org/10.1126/science.abe1530>.
- Doelman, J.C., Stehfest, E., Tabeau, A., van Meijl, H., 2019. Making the Paris agreement climate targets consistent with food security objectives. *Global Food Secur.* 23, 93–103. <https://doi.org/10.1016/j.gfs.2019.04.003>.

- Doelman, J.C., Stehfest, E., Tabeau, A., van Meijl, H., Lassaletta, L., Gernaat, D.E.H.J., Hermans, K., Harmsen, M., Daioglou, V., Biemans, H., van der Sluis, S., van Vuuren, D.P., 2018. Exploring SSP land-use dynamics using the IMAGE model: Regional and gridded scenarios of land-use change and land-based climate change mitigation. *Global Environ. Change* 48, 119–135. <https://doi.org/10.1016/j.gloenvcha.2017.11.014>.
- Downing, A.S., Bhowmik, A., Collste, D., Cornell, S.E., Donges, J., Fetzer, I., Häyhä, T., Hinton, J., Lade, S., Mooij, W.M., 2019. Matching scope, purpose and uses of planetary boundaries science. *Environ. Res. Lett.* 14 (7), 073005. <https://doi.org/10.1088/1748-9326/ab22c9>.
- Engel, Ernst, 1857. Die Productions- und Consumtionsverhältnisse des Königreichs Sachsen. Zeitschrift des statistischen Bureaus des Königlich Sächsischen Ministerium des Inneren.
- Faith, D.P., Ferrier, S., Williams, K.J., 2008. Getting biodiversity intactness indices right: ensuring that 'biodiversity' reflects 'diversity'. *Global Change Biol.* 14, 207–217. <https://doi.org/10.1111/j.1365-2486.2007.01500>.
- Francois, J., Van Meijl, H., Van Tongeren, F., 2005. Trade liberalization in the Doha development round. *Econ. Policy* 20, 350–391. <https://doi.org/10.1111/j.1468-0327.2005.00141>.
- Galaz, V., Biermann, F., Folke, C., Nielsson, M., Olsson, P., 2012. Global environmental governance and planetary boundaries: an introduction. *Ecol. Econ.* 81, 1–3. <https://doi.org/10.1016/j.ecolecon.2012.02.023>.
- Häyhä, T., Lucas, P.L., van Vuuren, D.P., Cornell, S.E., Hoff, H., 2016. From Planetary Boundaries to national fair shares of the global safe operating space — how can the scales be bridged? *Global Environ. Change* 40, 60–72. <https://doi.org/10.1016/j.gloenvcha.2016.06.008>.
- Heck, V., Hoff, H., Wirsensius, S., Meyer, C., Kreft, H., 2018. Land use options for staying within the Planetary Boundaries — Synergies and trade-offs between global and local sustainability goals. *Global Environ. Change* 49, 73–84. <https://doi.org/10.1016/j.gloenvcha.2018.02.004>.
- Hertel, T., & van der Mensbrugghe, D. (2016). Chapter 14: Behavioral Parameters (Center for Global Trade Analysis). Purdue University, West Lafayette, IN: Global Trade Analysis Project (GTAP). Retrieved from <[https://www.gtap.agecon.purdue.edu/resources/res\\_display.asp?RecordID=5138](https://www.gtap.agecon.purdue.edu/resources/res_display.asp?RecordID=5138)>.
- Hof, C., Voskamp, A., Biber, M.F., Böhning-Gaese, K., Engelhardt, E.K., Niamir, A., Willis, S.G., Hickler, T., 2018. Bioenergy cropland expansion may offset positive effects of climate change mitigation for global vertebrate diversity. *PNAS* 115 (52), 13294–13299. <https://doi.org/10.1073/pnas.1807745115>.
- Humpenöder, F., Popp, A., Bodirsky, B.L., Weindl, I., Biewald, A., Lotze-Campen, H., Dietrich, J.P., Klein, D., Kreyenweis, U., Müller, C., Rolinski, S., Stevanovic, M., 2018. Large-scale bioenergy production: how to resolve sustainability trade-offs? *Environ. Res. Lett.* 13 (2), 024011. <https://doi.org/10.1088/1748-9326/aa9e3b>.
- IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services), 2019. Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Bonn, Germany.
- Leclère, D., Obersteiner, M., Barrett, M., Butchart, S.H.M., Chaudhary, A., De Palma, A., DeClerck, F.A.J., Di Marco, M., Doelman, J.C., Dirauer, M., Freeman, R., Harfoot, M., Hasegawa, T., Hellweg, S., Hilbers, J.P., Hill, S.L.L., Humpenöder, F., Jennings, N., Krisztin, T., Mace, G.M., Ohashi, H., Popp, A., Purvis, A., Schipper, A.M., Tabeau, A., Valin, H., van Meijl, H., van Zeist, W.-J., Visconti, P., Alkemade, R., Almond, R., Bunting, G., Burgess, N.D., Cornell, S.E., Di Fulvio, F., Ferrier, S., Fritz, S., Fujimori, S., Grooten, M., Harwood, T., Havlik, P., Herrero, M., Hoskins, A. J., Jung, M., Kram, T., Lotze-Campen, H., Matsui, T., Meyer, C., Nel, D., Newbold, T., Schmidt-Traub, G., Stehfest, E., Strassburg, B.B.N., van Vuuren, D.P., Ware, C., Watson, J.E.M., Wu, W., Young, L., 2020. Bending the curve of terrestrial biodiversity needs an integrated strategy. *Nature* 585, 551–556. <https://doi.org/10.1038/s41586-020-2705-y>.
- Lenzen, M., Moran, D., Kanemoto, K., Foran, B., Lobefaro, L., Geschke, A., 2012. International trade drives biodiversity threats in developing nations. *Nature* 486 (7401), 109–112. <https://doi.org/10.1038/nature11145>.
- Leontief, W.W., 1936. Quantitative input and output relations in the economic systems of the United States. *Rev. Econ. Stat.* 18 (3), 105. <https://doi.org/10.2307/1927837>.
- Lucas, P.L., Wilting, H.C., Hof, A.F., van Vuuren, D.P., 2020. Allocating planetary boundaries to large economies: distributional consequences of alternative perspectives on distributive fairness. *Global Environ. Change* 60, 102017. <https://doi.org/10.1016/j.gloenvcha.2019.102017>.
- Marquardt, S.G., Guindon, M., Wilting, H.C., Steinmann, Z.J.N., Sim, S., Kulak, M., Huijbregts, M.A.J., 2019. Consumption-based biodiversity footprints — do different indicators yield different results? *Ecol. Ind.* 103, 461–470. <https://doi.org/10.1016/j.ecolind.2019.04.022>.
- Marques, A., Martins, I.S., Kastner, T., Plutzer, C., Theurl, M.C., Eisenmenger, N., Huijbregts, M.A.J., Wood, R., Stadler, K., Bruckner, M., Canelas, J., Hilbers, J.P., Tukker, A., Erb, K., Pereira, H.M., 2019. Increasing impacts of land use on biodiversity and carbon sequestration driven by population and economic growth. *Nat. Ecol. Evol.* 3 (4), 628–637. <https://doi.org/10.1038/s41559-019-0824-3>.
- Maxwell, S.L., Fuller, R.A., Brooks, T.M., Watson, J.E.M., 2016. Biodiversity: the ravages of guns, nets and bulldozers. *Nature News* 536 (7615), 143–145. <https://doi.org/10.1038/536143a>.
- Miller, R.E., Blair, P.M., 2009. *Input-Output Analysis: Foundations and Extensions*. Cambridge Univ. Press, Cambridge.
- Moran, D., Kanemoto, K., 2017. Identifying species threat hotspots from global supply chains. *Nat. Ecol. Evol.* 1, 0023. <https://doi.org/10.1038/s41559-016-0023>.
- Moran, D., Peterson, M., Veronesi, F., 2016. On the suitability of input-output analysis for calculating product-specific biodiversity footprints. *Ecol. Ind.* 60, 192–201. <https://doi.org/10.1016/j.ecolind.2015.06.015>.
- Müller, C., Elliott, J., Chryssanthacopoulos, J., Deryng, D., Folberth, C., Pugh, T.A.M., Schmid, E., 2015. Implications of climate mitigation for future agricultural production. *Environ. Res. Lett.* 10 (12), 125004. <https://doi.org/10.1088/1748-9326/10/12/125004>.
- Müller, C., Stehfest, E., van Minnen, J.G., Strengers, B., von Bloh, W., Beusen, A.H.W., Schaphoff, S., Kram, T., Lucht, W., 2016. Drivers and patterns of land biosphere carbon balance reversal. *Environ. Res. Lett.* 11 (4), 044002. <https://doi.org/10.1088/1748-9326/11/4/044002>.
- Nelson, G.C., Valin, H., Sands, R.D., Havlik, P., Ahammad, H., Deryng, D., Elliott, J., Fujimori, S., Hasegawa, T., Heyhoe, E., Kyle, P., Von Lampe, M., Lotze-Campen, H., Mason d'Croz, D., van Meijl, H., van der Mensbrugghe, D., Müller, C., Popp, A., Robertson, R., Robinson, S., Schmid, E., Schmitz, C., Tabeau, A., Willenbockel, D., 2014. Climate change effects on agriculture: economic responses to biophysical shocks. *PNAS* 111 (9), 3274–3279. <https://doi.org/10.1073/pnas.1222465110>.
- Newbold, T., Hudson, L.N., Arnell, A.P., Contu, S., De Palma, A., Ferrier, S., Hill, S.L.L., Hoskins, A.J., Lysenko, I., Phillips, H.R.P., Burton, V.J., Chng, C.W.T., Emerson, S., Gao, D.I., Pask-Hale, G., Hutton, J., Jung, M., Sanchez-Ortiz, K., Simmons, B.I., Whitmee, S., Zhang, H., Scharlemann, J.P.W., Purvis, A., 2016. Has land use pushed terrestrial biodiversity beyond the planetary boundary? A global assessment. *Science* 353 (6296), 288–291. <https://doi.org/10.1126/science.aaf2201>.
- Nykvist, B., Persson, A., Moberg, F., Persson, L., Cornell, S., Rockström, J., 2013. *National Environmental Performance on Planetary Boundaries: A Study for the Swedish Environmental Protection Agency* (No. 6576). Swedish Environmental Protection Agency, Stockholm.
- O'Neill, B.C., Krieglner, E., Ebi, K.L., Kemp-Benedict, E., Riahi, K., Rothman, D.S., van Ruijven, B.J., van Vuuren, D.P., Birkmann, J., Kok, K., Levy, M., Solecki, W., 2017. The roads ahead: Narratives for shared socioeconomic pathways describing world futures in the 21st century. *Global Environ. Change* 42, 169–180. <https://doi.org/10.1016/j.gloenvcha.2015.01.004>.
- O'Neill, D.W., Fanning, A.L., Lamb, W.F., Steinberger, J.K., 2018. A good life for all within planetary boundaries. *Nat. Sustainability* 1 (2), 88–95. <https://doi.org/10.1038/s41893-018-0021-4>.
- Overmars, K.P., Schulp, C.J.E., Alkemade, R., Verburg, P.H., Omtzigt, N., Schaminée, J. H.J., 2014. Developing a methodology for a species-based and spatially explicit indicator for biodiversity on agricultural land in the EU. *Ecological Indicators* 37, 186–198. <https://doi.org/10.1016/j.ecolind.2012.11.006>.
- Peters, G.P., Andrew, R., Lennox, J., 2011. Constructing an environmentally-extended multi-regional input-output table using the Gtap database. *Econ. Syst. Res.* 23 (2), 131–152. <https://doi.org/10.1080/09535314.2011.563234>.
- Popp, A., Calvin, K., Fujimori, S., Havlik, P., Humpenöder, F., Stehfest, E., Bodirsky, B.L., Dietrich, J.P., Doelmann, J.C., Gusti, M., Hasegawa, T., Kyle, P., Obersteiner, M., Tabeau, A., Takahashi, K., Valin, H., Waldhoff, S., Weindl, I., Wise, M., Krieglner, E., Lotze-Campen, H., Fricko, O., Riahi, K., van Vuuren, D.P., 2017. Land-use futures in the shared socio-economic pathways. *Global Environ. Change* 42, 331–345. <https://doi.org/10.1016/j.gloenvcha.2016.10.002>.
- Powers, R.P., Jetz, W., 2019. Global habitat loss and extinction risk of terrestrial vertebrates under future land-use-change scenarios. *Nat. Clim. Change* 9 (4), 323–329. <https://doi.org/10.1038/s41558-019-0406-z>.
- Ratnasiri, S., Bandara, J., 2017. Changing patterns of meat consumption and greenhouse gas emissions in Australia: Will kangaroo meat make a difference? *e0170130 PLoS ONE* 12 (2). <https://doi.org/10.1371/journal.pone.0170130>.
- Riahi, K., van Vuuren, D.P., Krieglner, E., Edmonds, J., O'Neill, B.C., Fujimori, S., Bauer, N., Calvin, K., Dellink, R., Fricko, O., Lutz, W., Popp, A., Cuarema, J.C., Kc, S., Leimbach, M., Jiang, L., Kram, T., Rao, S., Emmerling, J., Ebi, K., Hasegawa, T., Havlik, P., Humpenöder, F., Da Silva, L.A., Smith, S., Stehfest, E., Bosetti, V., Eom, J., Gernaat, D., Masui, T., Rogelj, J., Streffer, J., Drouet, L., Krey, V., Luderer, G., Harmsen, M., Takahashi, K., Baumstark, L., Doelman, J.C., Kainuma, M., Klimont, Z., Marangoni, G., Lotze-Campen, H., Obersteiner, M., Tabeau, A., Tavoni, M., 2017. The Shared Socioeconomic Pathways and their energy, land use, and greenhouse gas emissions implications: an overview. *Global Environ. Change* 42, 153–168. <https://doi.org/10.1016/j.gloenvcha.2016.05.009>.
- Rockström, J., Steffen, W., Noone, K., Persson, H., Chapin, F.S., Lambin, E., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J., 2009. Planetary boundaries: exploring the safe operating space for humanity. *Ecol. Soc.* 14.
- Rosenzweig, C., Elliott, J., Deryng, D., Ruane, A.C., Müller, C., Arneth, A., Boote, K.J., Folberth, C., Glotter, M., Khabarov, N., Neumann, K., Piontek, F., Pugh, T.A.M., Schmid, E., Stehfest, E., Yang, H., Jones, J.W., 2014. Assessing agricultural risks of climate change in the 21st century in a global gridded crop model intercomparison. *PNAS* 111 (9), 3268–3273. <https://doi.org/10.1073/pnas.1222463110>.
- Ryberg, M.W., Owsianiak, M., Clavreul, J., Mueller, C., Sim, S., King, H., Hauschild, M.Z., 2018. How to bring absolute sustainability into decision-making: an industry case study using a Planetary Boundary-based methodology. *Sci. Tot. Environ.* 634, 1406–1416. <https://doi.org/10.1016/j.scitotenv.2018.04.075>.
- Schipper, A.M., Belmaker, J., Miranda, M.D., Navarro, L.M., Böhning-Gaese, K., Costello, M.J., Dornelas, M., Foppen, R., Hortal, J., Huijbregts, M.A.J., Martín-López, B., Pettorelli, N., Queiroz, C., Rossberg, A.G., Santini, L., Schiffers, K., Steinmann, Z.J.N., Visconti, P., Rondinini, C., Pereira, H.M., 2016. Contrasting changes in the abundance and diversity of North American bird assemblages from 1971 to 2010. *Glob. Change Biol.* 22 (12), 3948–3959. <https://doi.org/10.1111/gcb.2016.22.issue-1210.1111/gcb.13292>.
- Schipper, A.M., Hilbers, J.P., Meijer, J.R., Antão, L.H., Benítez-López, A., Jonge, M.M.J., Leemans, L.H., Scheper, E., Alkemade, R., Doelman, J.C., Mylius, S., Stehfest, E., Vuuren, D.P., Zeist, W.-J., Huijbregts, M.A.J., 2020. Projecting terrestrial

- biodiversity intactness with GLOBIO 4. *Glob. Change Biol.* 26 (2), 760–771. <https://doi.org/10.1111/gcb.v26.210.1111/gcb.14848>.
- Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., de Vries, W., de Wit, C.A., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B., Sörlin, S., 2015. Planetary boundaries: guiding human development on a changing planet. *Science*. <https://doi.org/10.1126/science.1259855>, 1259855.
- Stehfest, E., Bouwman, L., van Vuuren, D.P., den Elzen, M.G.J., Eickhout, B., Kabat, P., 2009. Climate benefits of changing diet. *Clim. Change* 95 (1–2), 83–102. <https://doi.org/10.1007/s10584-008-9534-6>.
- Stehfest, E., van Vuuren, D., Bouwman, L., Kram, T., 2014. *Integrated Assessment of Global Environmental Change with IMAGE 3.0: Model Description and Policy Applications*. Netherlands Environmental Assessment Agency (PBL).
- Stehfest, E., van Zeist, W.-J., Valin, H., Havlik, P., Popp, A., Kyle, P., Tabeau, A., Mason-D'Croz, D., Hasegawa, T., Bodirsky, B.L., Calvin, K., Doelman, J.C., Fujimori, S., Humpenöder, F., Lotze-Campen, H., van Meijl, H., Wiebe, K., 2019. Key determinants of global land-use projections. *Nat. Commun.* 10, 1–10. <https://doi.org/10.1038/s41467-019-09945-w>.
- Temme, A.J.A.M., Verburg, P.H., 2011. Mapping and modelling of changes in agricultural intensity in Europe. *Agriculture, Ecosystems & Environment* 140, 46–56. <https://doi.org/10.1016/j.agee.2010.11.010>.
- UN (United Nations), 2015. *Transforming Our World: The 2030 Agenda for Sustainable Development*. Resolution adopted by the General Assembly. (A/RES/70/1, 21 October).
- Vačkár, D., ten Brink, B., Loh, J., Baillie, J.E.M., Reyers, B., 2012. Review of multispecies indices for monitoring human impacts on biodiversity. *Ecol. Indic.* 17, 58–67. <https://doi.org/10.1016/j.ecolind.2011.04.024>.
- van Meijl, H., Van Tongeren, F., 2004. Projections of the Chinese economy to 2020: the impact of agricultural and trade policies and implications for global trade. In: *China's Food Economy in the Early 21st Century*. Report, 6(04).
- van Vuuren, D.P., Kok, M.T.J., Girod, B., Lucas, P.L., de Vries, B., 2012. Scenarios in Global Environmental Assessments: key characteristics and lessons for future use. *Global Environ. Change* 22 (4), 884–895. <https://doi.org/10.1016/j.gloenvcha.2012.06.001>.
- van Vuuren, D.P., Stehfest, E., Gernaat, D.E.H.J., Doelman, J.C., van den Berg, M., Harmsen, M., de Boer, H.S., Bouwman, L.F., Daioglou, V., Edelenbosch, O.Y., Girod, B., Kram, T., Lassaletta, L., Lucas, P.L., van Meijl, H., Müller, C., van Ruijven, B.J., van der Sluis, S., Tabeau, A., 2017. Energy, land-use and greenhouse gas emissions trajectories under a green growth paradigm. *Global Environ. Change* 42, 237–250. <https://doi.org/10.1016/j.gloenvcha.2016.05.008>.
- van Vuuren, D.P., Stehfest, E., Gernaat, D.E.H.J., van den Berg, M., Bijl, D.L., de Boer, H. S., Daioglou, V., Doelman, J.C., Edelenbosch, O.Y., Harmsen, M., Hof, A.F., van Sluisveld, M.A.E., 2018. Alternative pathways to the 1.5 °C target reduce the need for negative emission technologies. *Nat. Clim. Change* 8, 391–397. <https://doi.org/10.1038/s41558-018-0119-8>.
- van Zeist, W.-J., Stehfest, E., Doelman, J.C., Valin, H., Calvin, K., Fujimori, S., Hasegawa, T., Havlik, P., Humpenöder, F., Kyle, P., Lotze-Campen, H., Mason-D'Croz, D., van Meijl, H., Popp, A., Sulser, T.B., Tabeau, A., Verhagen, W., Wiebe, K., 2020. Are scenario projections overly optimistic about future yield progress? *Global Environ. Change* 64, 102120. <https://doi.org/10.1016/j.gloenvcha.2020.102120>.
- Veronesi, F., Moran, D., Stadler, K., Kanemoto, K., Wood, R., 2017. Resource footprints and their ecosystem consequences. *Sci. Rep.* 7 (1) <https://doi.org/10.1038/srep40743>.
- Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., Garnett, T., Tilman, D., DeClerck, F., Wood, A., Jonell, M., Clark, M., Gordon, L.J., Fanzo, J., Hawkes, C., Zurayk, R., Rivera, J.A., De Vries, W., Majele Sibanda, L., Afshin, A., Chaudhary, A., Herrero, M., Agustina, R., Branca, F., Lartey, A., Fan, S., Crona, B., Fox, E., Bignet, V., Troell, M., Lindahl, T., Singh, S., Cornell, S.E., Srinath Reddy, K., Narain, S., Nishtar, S., Murray, C.J.L., 2019. Food in the Anthropocene: the EAT–Lancet Commission on healthy diets from sustainable food systems. *The Lancet* 393 (10170), 447–492. [https://doi.org/10.1016/S0140-6736\(18\)31788-4](https://doi.org/10.1016/S0140-6736(18)31788-4).
- Wilting, H.C., Schipper, A.M., Bakkenes, M., Meijer, J.R., Huijbregts, M.A.J., 2017. Quantifying biodiversity losses due to human consumption: a global-scale footprint analysis. *Environ. Sci. Technol.* 51 (6), 3298–3306. <https://doi.org/10.1021/acs.est.6b05296>.
- Woltjer, G.B., Kuiper, M., Kavallari, A., Van Meijl, H., Powell, J., Rutten, M., Shutes, L., Tabeau, A., 2014. *The Magnet Model – Module description*. LEI, part of Wageningen University and Research Centre, The Hague.