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Shared and environmentally just responsibility for global biodiversity loss

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A R T I C L E I N F O

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A B S T R A C T

Human land use is the main driver of terrestrial biodiversity loss. It has been argued that producers and consumers have a shared responsibility for biodiversity loss because this land use is directly and indirectly driven by the local and global demand for products. Such responsibility sharing would be an important step for global biodiversity cooperation and conservation. Here, we use a global multiregional input-output framework to estimate consumption-based biodiversity loss, integrating with both the physical Food and Agriculture Biomass Input-Output (FABIO) dataset and a global monetary input-output table (EXIOBASE). We use an environmental justice framework for assigning biodiversity loss responsibility between producers and consumers. In this framework, we employ the Human Development Index (HDI) as a proxy of the weighting parameter for both producers and consumers. An environmental justice perspective may provide a fairer distribution of responsibility in a world where different nations have very different capabilities and see varying benefits from international trade. Environmentally just accounting increases the footprint of the Global North compared to other common approaches for sharing responsibility across all producers and consumers along international supply chains. We describe how environmental justice may inform cooperation in biodiversity protection between stakeholders along global supply chains.

1. Introduction

Environmental inequalities threaten health and wellbeing, economic development, and social cohesion (Mohai et al., 2009; Sze and London, 2008). Environmental justice is a multidisciplinary concept that aims to address these inequalities and plays an important role in sustainable development. Indeed, some studies suggest that environmental and economic inequalities are likely to be the largest barriers to sustainability (Adger, 2002; Liu, 2018). Local studies on environmental justice are most common when assessing differences in pollution exposures between people of different races, or incomes (Banzhaf et al., 2019; Mohai et al., 2009; Sze and London, 2008). But there are also significant environmental justice issues at a larger scale (Mohai et al., 2009; Sze and London, 2008), for example, biodiversity loss due to land use across the world is often linked to demand across a globalized trade system (Boillat et al., 2020). In a particularly appropriate example, many developed countries (e.g. the US, Japan, and Germany) and some developing countries (e.g. China and India) have spared land for domestic afforestation but at the same time import products that drive deforestation elsewhere (e.g. Brazil and Indonesia) (Hoang and Kanemoto, 2021). The example depicts the concept of telecoupling that describes how socioeconomic and environmental impacts are connected across large distances (Liu et al., 2013). Given the fact that trade networks can embed economic and environmental inequalities, some researchers have suggested telecoupling studies should incorporate environmental justice perspectives wherever possible (Boillat et al., 2020).

Prominent environmental justice frameworks cover four dimensions: distributive justice, recognitional justice, procedural justice, and concepts of capabilities (Menton et al., 2020). Telecoupling analysis is needed for a full accounting of all these dimensions. For instance, Boillat et al. (2020) suggest that distributive justice would help identify environmental winners and losers in trade, which could then be used to provide aid for procedural and recognitional justice by explicitly integrating responsibility and agent perspectives. The final dimension, the capabilities approach, is also important, as it helps to determine the ability of agents to shoulder responsibilities in telecoupled systems, such as in international trade (Menton et al., 2020).

Environmental responsibility must in some way be attributed to

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associated stakeholders, such as producers and consumers, along global supply chains. One option is to place responsibilities with a single stakeholder; however, this could be considered unfair, since producers generate environmental pressures, income receivers enable them, final consumers drive them, and all actors benefit from them through the production of added value (Steininger et al., 2016; Tukker et al., 2020). That is, all stakeholders stand to gain from the activity along supply chains. Various approaches for 'sharing responsibilities' between producers and consumers have been proposed, including production-based, consumption-based, income-based, value-added-based responsibility and a mix of these four principles (Tukker et al., 2020). The production-based approach allocates full responsibility to producers of goods and services where environmental pressures occur (Tukker et al., 2020). However, nations could use this to transfer production and related environmental pressures overseas to other jurisdictions (an example of this is carbon leakage). The recognition of this shortcoming is addressed by a consumption-based approach that allocates full responsibility of environmental pressures generated upstream along supply chains to final consumers (Davis and Caldeira, 2010; Hertwich and Peters, 2009). However, the consumption-based approach in turn leaves the producers and exporters without any responsibility. A way to address this is to use an income-based approach that attributes some responsibilities to any agent along supply chains who receive income via wages and/or capital return, from the initial producer of primary goods to suppliers of intermediate goods and services (Liang et al., 2017; Marques et al., 2012). However, the income-based approach does not assign any responsibility for the pressures embodied in imported commodities consumed in final demand (i.e. the final consumers) (Marques et al., 2012). Finally, a similar, value-added approach allocates responsibility to producers of value added. The value-added approach first allocates all environmental pressures downstream of a primary product to the final products, and then re-allocates them to all agents along each product's supply chain based on value-added at each stage (Piñero et al., 2019). Neither the income-based nor the value-added approach places any responsibility on the final consumers importing the final goods, so final consumers will see no responsibility for their consumption choices.

Some studies have used a mix of two or more of these approaches, such as the average of consumption- and income-based approaches (Qian et al., 2019; Rodrigues et al., 2006), the average of production- and consumption-based accounting weighted by border carbon taxation (Chang, 2013), or the average of production- and income-based footprints (Gallego and Lenzen, 2005). Jakob et al. (2021) argued that these approaches do not have a solid theoretical foundation and proposed an 'economic benefit shared responsibility' approach that allocates environmental pressures between producers and consumers based on the difference between their willingness to pay and the market price (economic surplus) (Jakob et al., 2021). Indeed, all the sharing methods described thus far focus solely on economic factors (such as value-added or income) and do not a priori include environmental justice concerns. Oliveira (2019) suggests a framework to allocate environmental responsibility to both producers and consumers using a weighting that depends on development and justice parameters (Oliveira, 2019), thereby addressing distributive justice and the concept of capabilities. This framework has the benefit of broadening the concept of environmental responsibility by incorporating a variety of social, political, and economic aspects.

Studies on responsibility sharing have generally focused on greenhouse gas emissions (GHGs) (e.g. Jakob et al., 2021; Steininger et al., 2016; Tukker et al., 2020). As yet, there have been limited studies on other important environmental pressures, such as biodiversity loss due to land use. Biodiversity is under severe threat and is declining at an unprecedented rate in human history (IPBES, 2019). Increasing levels of human-driven intensive land use is the largest threat to terrestrial biodiversity and has caused severe global species loss (Díaz et al., 2019; Ellis et al., 2021). There is an increasing appreciation that biodiversity loss has an adverse impact on ecosystem services (Cardinale et al., 2012). Some efforts have been made to protect global biodiversity, for example, the 2010 Aichi targets in the UN Convention on Biological Diversity (CBD, 2010; Díaz et al., 2019). However, only 6 of 20 Aichi targets were partially fulfilled and none were fully met (CBD, 2020; Díaz et al., 2019). This may be partly due to a lack of responsibility sharing for biodiversity protection efforts between nations. While these global biodiversity protection efforts have generally failed, the ambition of future goals has been raised in the post-2020 biodiversity framework (CBD, 2020; Díaz et al., 2020). Achieving these goals in an environmentally just way will require responsibility assessments of biodiversity loss of nations.

This study aims to allocate biodiversity loss to both producers and consumers, considering environmental justice. To do so, we use the human development index (HDI) as a proxy for the countries' capabilities. Thereby, we broaden the perspective of responsibility-sharing from a solely economic assessment. The results can provide policy guidance for reducing biodiversity loss and may aid in the achievement of sustainable development goals surrounding biodiversity (such as SDG15 on *protecting terrestrial ecosystems and halting biodiversity loss*).

2. Methods

In this study, we employ the footprint framework based on environmentally just accounting proposed by Oliveira (2019) and operationalize it by choosing the Human Development Index (HDI) as a weighting parameter (Section 2.1). We choose HDI as the weighting parameter since it is a relatively transparent indicator that reflects several different justice metrics. We use a global, environmentally extended, multiregional input-output (MRIO) approach that has been widely used to allocate social and environmental responsibilities between actors along global supply chains (Jakob et al., 2021; Sun et al., 2020, 2019; Tukker et al., 2020; Wiedmann and Lenzen, 2018). We integrate the physical Food and Agriculture Biomass Input-Output (FABIO) dataset with a global monetary input-output table (EXIOBASE) (Section 2.2) to capture all economic sectors driving biodiversity loss. To assess biodiversity loss, we apply characterization factors (CFs) representing *global species-equivalents potentially lost (hereafter referred to as species lost) per* m^2 *land use* (Chaudhary and Brooks, 2018) with land use areas for five land use types (cropland, pasture, managed forest, planted forest, and urban land) and three land use intensities (minimal, light, and intense) (Sections 2.3 and 2.4).

2.1. Responsibility Sharing

The environmentally just accounting aims to capture environmental justice in responsibility sharing. The critical step in this method is to construct a weighting between producer- and consumer-perspectives to reallocate responsibility. Oliveira (2019) demonstrates this for two regions and we extend this here for multiple regions, giving:

$$\boldsymbol{F}_r^{just} = \boldsymbol{F}_r^{dom} + \sum_{s,s\neq r} \alpha_s \boldsymbol{F}_s^{imp} + \sum_{t,t\neq r} \beta_t \boldsymbol{F}_t^{exp} + \boldsymbol{F}_r^{fd} \quad (1)$$

Where $\boldsymbol{F}_r^{just}$ is the justice-based footprint in region r, $\boldsymbol{F}_r^{dom}$ gives environmental pressures related to domestic production for domestic consumption in region r, $\boldsymbol{F}_s^{imp}$ the environmental pressures embodied in imports into region r from region s, $\boldsymbol{F}_t^{exp}$ shows environmental pressures associated with production for export in region r to region t, $\boldsymbol{F}_r^{fd}$ gives environmental pressures on land use for domestic consumption activities only and is not involved in international trade (i.e. infrastructure land in the study) in region r. α and β are the weighting parameters, where

$$\alpha_s = \frac{C_r}{C_r + P_s} \quad (2)$$

$$\beta_t = \frac{P_r}{P_r + C_t} \tag{3}$$

C_r, P_r, C_t, and P_s are the normalized weighting indicators between consumption and production perspectives.

Weightings can be constructed in numerous ways (Oliveira, 2019), for example, for producers, as a function of (1) technological improvement capacity, (2) technological sectoral improvement capacity, and (3) availability of "greener" substitute goods and services. Similarly, the weighting parameters for consumers can be a function of (1) general environmental awareness of the population (related to education), (2) their purchase power (corrected by the inequality level), and (3) availability to consumers of "greener" substitute goods and services according to Oliveira (2019). However, she does not propose a specific index for application. In this paper, we distribute biodiversity loss responsibility among producers (P_S in eq. 5, and P_r in equation 6) and consumers (C_r in eq. 5, and C_t in eq. 6) based on the Human Development Index (HDI) for the respective countries, resulting in identical weighting factors α and β for each combination of two regions. The HDI is a geometric mean of the normalized life expectancy index (using life expectancy at birth), education index (using expected years of schooling for children and mean years of schooling for adults), and gross national income (GNI) index (using GNI per capita (PPP)). Nations with higher development have more capacity for greener technology innovation because they have both higher human capital (higher education with longer life expectancy) and economic capital (higher gross national income) (Cazzolla Gatti, 2016). Citizens in nations with higher development may also have higher environmental awareness and purchase power to buy "greener" substitute goods and services (Cazzolla Gatti, 2016).

2.2. MRIO Analysis

Environmental pressures along global supply chains can be calculated using MRIO analysis (for more details please see (Bruckner and Giljum, 2018; Sun et al., 2021)). We consider land use for food consumption ($\boldsymbol{y_{fabio}}$) and non-food consumption ($\boldsymbol{y_{exio}}$) separately, based on an integration of the Food and Agriculture Biomass Input–Output model (FABIO) and the monetary economic MRIO EXIOBASE. The integration of FABIO and EXIOBASE datasets allows users to trace environmental pressures embodied in international trade at unprecedented product detail. FABIO is a consistent, balanced, physical input–output database based on FAOSTAT data, covering 191 countries and 128 agriculture, food, and forestry products from 1986 to 2013. For further information on its construction see Bruckner et al. (2019). However, FABIO does not cover other economic sectors. In order to improve coverage, we integrate the dataset with EXIOBASE v3.6, a highly detailed global multi-regional input-output database, including 200 products, 163 industries, individual countries for EU27 members, a further 17 economically larger countries and 5 rest of world regions (Stadler et al., 2018). The input-output relationship between economic sectors within or between countries for the integrated FABIO and EXIOBASE framework is depicted by a block matrix (i.e. A_{fabio}, A_{exio}, A_{other} in Table 1). A_{fabio} and A_{exio} describe the input-output relationship between economic sectors and nations for FABIO and EXIOBASE, respectively. A_{other} is the matrix of technical coefficients linking the mostly agricultural products from FABIO to the non-agricultural products in EXIOBASE. The integrated FABIO and EXIOBASE framework assumes that the feedback from non-agricultural products in EXIOBASE to agricultural products in FABIO is zero. Since all production-based land use is linked to agricultural and forestry products (i.e. from FABIO to EXIOBASE), we do not need feedbacks from EXIOBASE to FABIO in this use case. We use FABIO and EXIOBASE v3.6 for the year 2005. Estimating the spatial distribution of consumption-based land use based on MRIO analysis is expressed mathematically in eqs. (4, 5).

Table 1
Description of variables and parameters in MRIO analysis.

Names	Description
$\boldsymbol{F^s}$	The global spatial distribution of land use driven by final consumption of country *s* for both FABIO and EXIOBASE.
R^r	The spatial distribution, represented in absolute values, of land use in country *r*.
e_i^r	The environmental intensity of product *i* in the producing country *r*.
$\boldsymbol{y}^{ts}{}_{fabio,j}$	The final consumption of FABIO product *j* in country *s* that originates from country *t*, which is the last country exporting to country *s* in FABIO.
$\boldsymbol{y}^{uv}{}_{exio,k}$	The final consumption of EXIOBASE product *k* in country *v* that originates from country *u*, which is the last country exporting to country *u* in the other-uses matrix ($\boldsymbol{A_{other}}$).
$d^r{}_i$	The total land use of product *i* in country *r*.
$HH^s{}_i$	The direct land use driven by domestic consumption of product *i* in country *s*.
$\boldsymbol{L}$	The Leontief inverse matrix; $\boldsymbol{L_A}$, $\boldsymbol{L_B}$, $\boldsymbol{L_D}$ are the subcomponents of L. The subscripts *i, j, k* stand for products, and the superscripts *r, t, u* represent countries, as for the above variables.
$\boldsymbol{I_{fabio}}$	The identity matrix with the same dimension as FABIO.
$\boldsymbol{I_{exio}}$	The identity matrix with the same dimension as EXIOBASE.
$\boldsymbol{A_{fabio}}$	The technical matrix of FABIO.
$\boldsymbol{A_{exio}}$	The technical matrix of EXIOBASE.
$\boldsymbol{A_{other}}$	The technical matrix linking the agricultural products from FABIO to the non-agricultural products in EXIOBASE.

$$\boldsymbol{F}^s = \sum_{i,r} R^r \frac{\boldsymbol{e}_i^r \sum_{jt} \boldsymbol{L_A}{}_{ij}^{rt} \boldsymbol{y}_{fabio,j}^{ts}}{d_i^r} + \sum_{i,r} R^r \frac{\boldsymbol{e}_i^r \sum_{jt} \boldsymbol{L_B}{}_{ik}^{ru} \boldsymbol{y}_{exio,k}^{uv}}{d_i^r} + \sum_i R^s \frac{\sum_i HH_i^s}{d_i^s} \tag{4}$$

$$\boldsymbol{L} = \begin{pmatrix} (\boldsymbol{I}_{fabio} - \boldsymbol{A}_{fabio})^{-1} & (\boldsymbol{I}_{fabio} - \boldsymbol{A}_{fabio})^{-1} \boldsymbol{A}_{other} (\boldsymbol{I}_{exio} - \boldsymbol{A}_{exio})^{-1} \\ \boldsymbol{0} & (\boldsymbol{I}_{exio} - \boldsymbol{A}_{exio})^{-1} \end{pmatrix} = \begin{pmatrix} \boldsymbol{L_A} & \boldsymbol{L_B} \\ \boldsymbol{0} & \boldsymbol{L_D} \end{pmatrix} \tag{5}$$

EXIOBASE has a higher spatial aggregation, i.e., the number of countries in FABIO and the number of countries in EXIOBASE are different. We assume the same per-capita consumption for FABIO countries that are in the five "rest of world" regions in EXIOBASE.

2.3. Land Use Datasets

The base year is 2005, as the CFs we use are based on land use types in 2005 (Hoskins et al., 2016). Land use intensity is derived from the Global Land System (van Asselen and Verburg, 2013, 2012). To keep the geographic data consistent, we aggregate all land use maps to a common resolution of 5 arcmin that is in line with the lowest resolution of spatial datasets in this study such as land use intensity and cropland. For cropland, we allocated national harvested area of 168 types of primary crops in FAOSTAT into grid cells based on 40 crop maps derived from the Spatial Production Allocation Model (SPAM) at 5 arcmin resolution (You et al., 2017). Cropland used to produce animal fodder is allocated analogously using aggregated fodder maps from EarthStat (Monfreda et al., 2008). For forestry, we used the latest, high-resolution global forest data, which distinguishes between different management and use patterns (Schulze et al., 2019). For infrastructure, we used ESA CCI land cover maps (using the *Urban Areas* classification at 300 m resolution) and aggregate to country level for the data required in the land use of final demand (ESA, 2017). For pasture, we used a high-resolution (30 arcsec) map from 2005 (Hoskins et al., 2016), excluded non-productive areas (aboveground NPP below 20 g C m^{-2} yr^{-1}) (Marques et al., 2019), and capped the pasture at 100% total land-use coverage in each grid cell.

2.4. Biodiversity Loss Associated with Land Use

The biodiversity characterization factors (CFs) we use allow for an estimation of global potential extinctions driven by per unit of land use

(Chaudhary and Brooks, 2018). That means that the species will go extinct globally if land use is maintained in the same way as in the base year. Different biodiversity models to assess biodiversity loss due to land use have advantages and disadvantages. For example, some other models can only estimate relative local biodiversity loss (Leclère et al., 2020). The CFs here consider five taxa (mammals, birds, amphibians, reptiles, and plants) and five land use types (managed forest, plantation, pasture, cropland, and urban) under three intensity levels (minimal, light, and intense use) for 804 terrestrial ecoregions (Table S3 in Chaudhary and Brooks, 2018). The CFs were derived from the countryside Species–Area Relationship (SAR) for regional species loss (Chaudhary and Brooks, 2018). The regional species loss multiplied with a vulnerability score of species gives the global species loss whose unit is *global species-equivalents potentially lost* (referenced to as species lost). After computing the spatial distribution per unit area of each land use type at different land use intensities driven by final consumption in a given region, we multiply the corresponding CFs with consumption-based land use data to obtain consumption-based global species loss for each taxon.

$$SL^{s}_{global,g,m,n} = \mathrm{CF}_{global,g,m,n} \times \mathrm{Area}^{s}_{m,n} \tag{6}$$

$SL^{s}_{global,g,m,n}$ is the number of species lost for each taxon g for a different land use type and intensity m in each grid cell n driven by final consumption in country s. $CF_{global,g,m,n}$ is the land occupation CF (species lost per unit land use) for taxon g at a different land use type and intensity m in each grid cell n. $Area^{s}_{m,n}$ is the land use for each different land use type and intensity m in each grid cell n driven by final consumption in country s from above MRIO analysis.

3. Results

The spatial distribution of biodiversity loss is highly uneven across the world (Fig. 1 A, D). Globally, there is a total potential loss from the production and trade analysed here of 5039 terrestrial plant species (hereafter referred to as plants) over the long term and 1765 terrestrial vertebrate species (hereafter referred to as vertebrates, including mammals, birds, amphibians, and reptiles) (Fig. A1 A, D). From a production-based perspective, the Global South (often equated with developing countries, see Table A1) sees a larger biodiversity loss with 4119 plant losses and 1604 vertebrate losses, compared to the Global North (often equated with developed countries, see Table A1) with 920 and 161, respectively. This is mainly due to the much larger population in the Global South, accounting for around 80% of the global total (United Nations, 2019) and seeing higher amounts of land use. From a consumption-based perspective, the Global North sees 1300 plant losses and 368 vertebrate losses, larger than its production-based footprint. The gap between consumption-based and production-based footprints is due to biodiversity loss embodied in traded commodities. Per-capita consumption-based plant and vertebrate loss are quite similar between

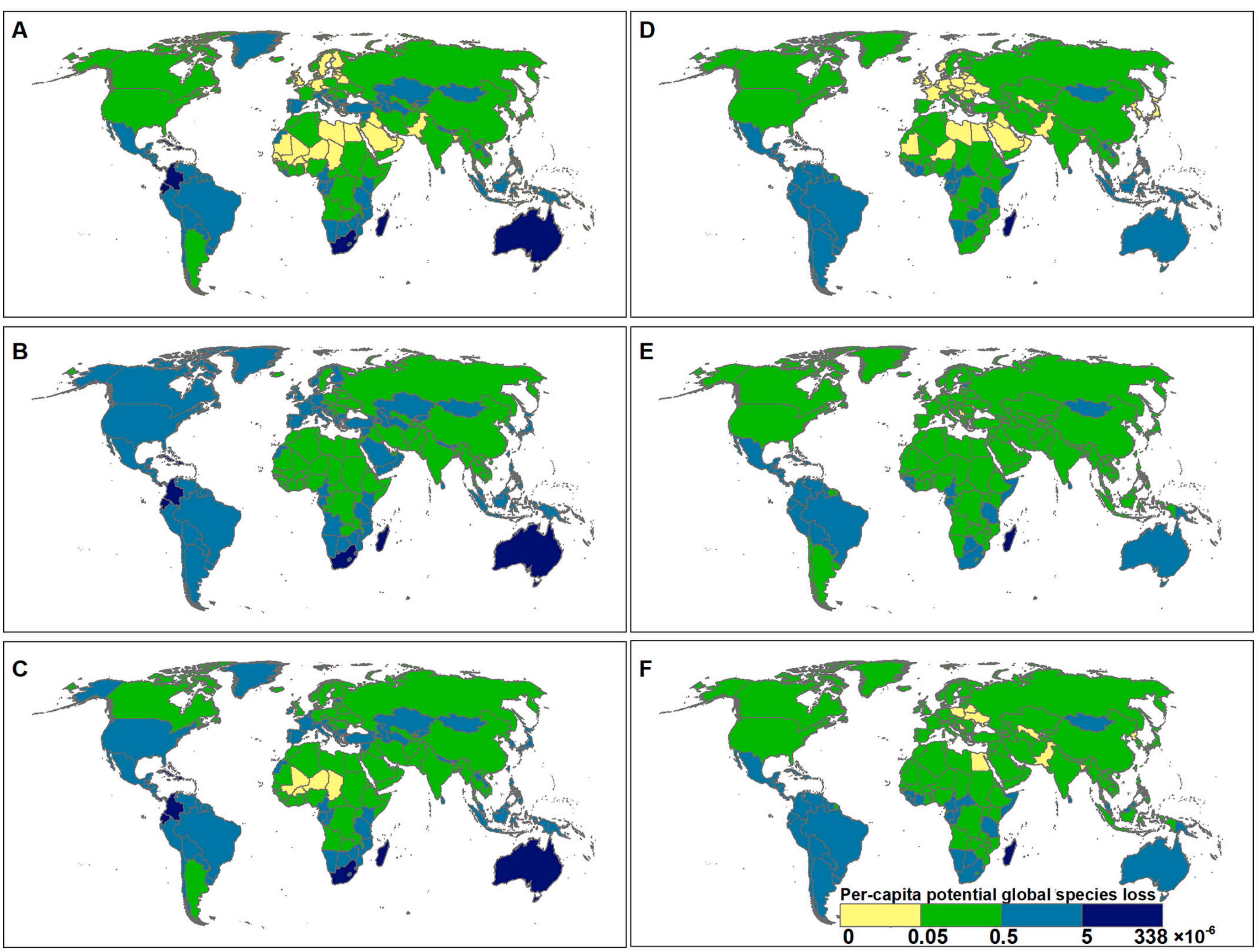

Fig. 1. Different responsibilities for per-capita potential global species loss from land use. A) production-based plant loss, B) consumption-based plant loss, C) justice-based plant loss, D) production-based vertebrate loss, E) consumption-based vertebrate loss, F) justice-based vertebrate loss.

the Global South and Global North at 0.7×10^{-6} and 0.3×10^{-6}, compared to 1.0×10^{-6} and 0.3×10^{-6}, respectively (Fig. 1 B, E). 56% and 73% of the consumption-based plant and vertebrate losses of the Global North is from imported products from all regions as opposed to domestically produced products, compared to 14% and 16% in the Global South. In terms of biodiversity loss flows from the Global South to the Global North, 40% and 63% of the consumption-based plant and vertebrate losses of the Global North is embodied in the imported commodities from the Global South.

Global biodiversity loss shows different patterns based on distinct responsibility sharing approaches (Fig. 1). There are two main types of regions with high justice-based biodiversity footprints: biodiverse regions, such as South Africa and Brazil as a result of domestic consumption; and large importers such as the US. For plant species, South Africa sees the largest loss from a production-based, consumption-based, and environmentally just perspective (597, 542, and 567 species lost, respectively). Mexico sees the second largest production-based and justice-based biodiversity loss, with around half of that lost in South Africa (293 and 285 species lost, respectively). In contrast, the US drives the second largest consumption-based biodiversity loss (323 species lost), mainly through imports from other nations. São Tomé and Príncipe sees the largest per-capita plant loss from a production- and consumption-based, and environmentally just perspective (338×10^{-6}, 324×10^{-6}, and 331×10^{-6} species lost per capita, respectively; Fig. 1, B), as São Tomé and Príncipe is an important region for endemic species with high CFs for land use (Chaudhary and Brooks, 2018; Jones, 1994). The next largest environmentally just and consumption-based biodiversity loss per capita is seen in Jamaica, with far fewer species loss at 20×10^{-6} and 19×10^{-6} species, respectively. Focusing on vertebrates specifically, Brazil sees the largest impacts from the production-based and environmentally just perspectives (157 and 135 species lost, respectively), while the US drives the largest consumption-based impacts (126 species lost). As with plant losses, São Tomé and Príncipe drives the largest per-capita vertebrate losses from all three perspectives, with 15×10^{-6}, 14×10^{-6}, and 14×10^{-6} species, respectively (production-based, consumption-based, and justice-based). Madagascar's land use causes the second largest per-capita vertebrate losses, with 7×10^{-6}, 5×10^{-6}, and 6×10^{-6} species for the three perspectives,

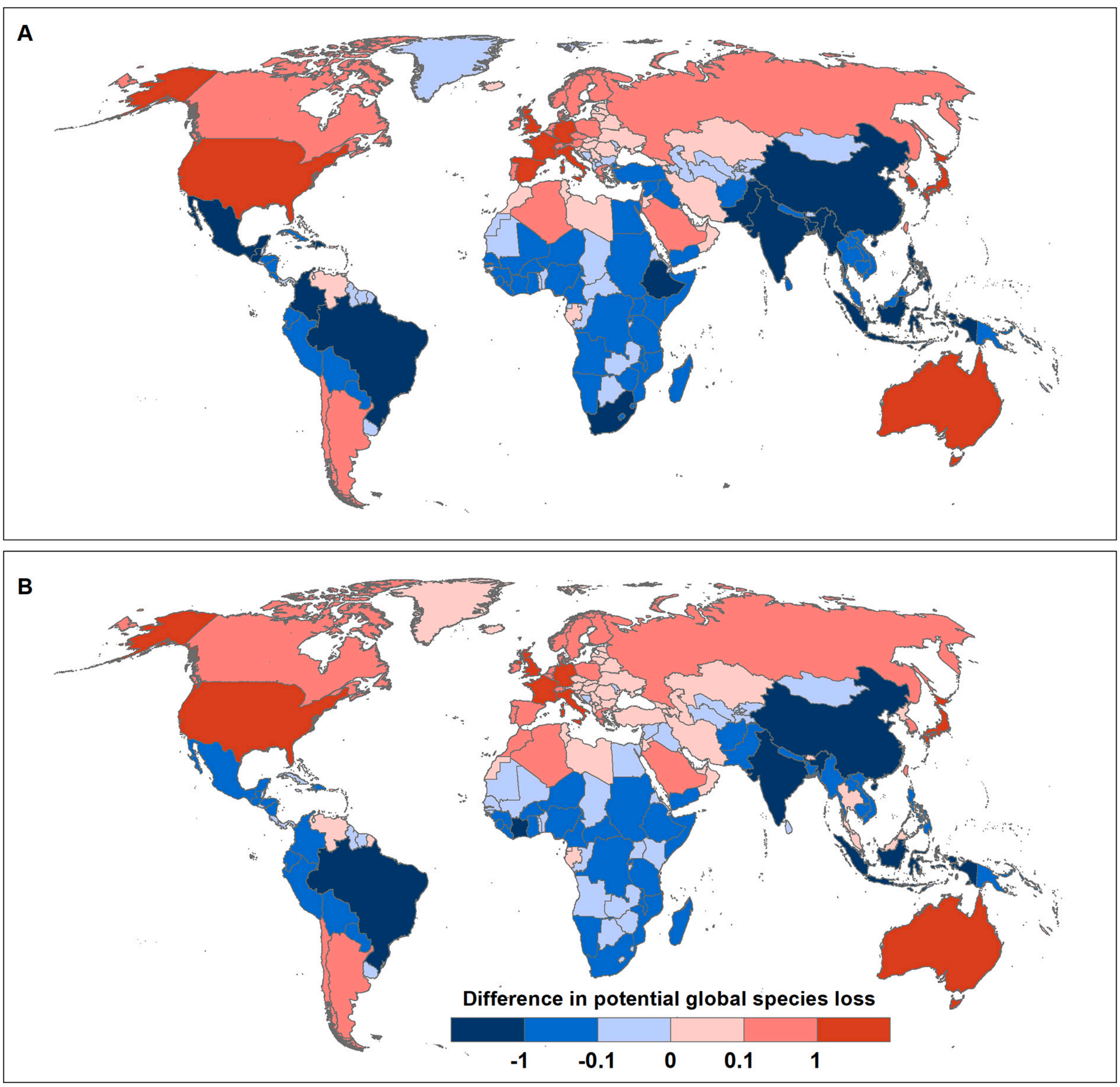

Fig. 2. The difference between national, justice-based footprints and the average of production- and consumption-based footprints of global species losses from land use for A) plants, and B) vertebrates (mammals, birds, amphibians, and reptiles). A negative value indicates higher biodiversity loss for an average of production- and consumption-based perspective, and a positive value indicates higher biodiversity loss for the justice-based perspective.

respectively.

The average of production- and consumption-based accounting is the simplest way to share responsibility between producers and consumers. The approach assumes equal responsibility for impacts embodied in international trade between producers and consumers, ignoring attributes of producers and consumers. We show the difference between the average of production- and consumption-based accounting and the justice-based accounting in Fig. 2. The justice-based accounting implies that nations with higher HDI shoulder more responsibility than their trade partners with lower HDI. As nations in the Global North have a higher HDI, with generally higher education and purchasing power, they typically import more biodiversity loss embodied in commodities from other regions. Nations in the Global North almost always shoulder more responsibility than their trade partners in the Global South. The consideration of environmental justice increases the responsibility of the Global North. For instance, the justice-based accounting of plant and vertebrate losses in the Global North are 48 and 21 species higher than the average of production- and consumption-based accounting. Looking at specific countries, the justice-based footprints of the United States (HDI of 0.90) and Australia (HDI of 0.91) are 13 and 9 species higher than their average of production- and consumption-based plant losses from land use.

To show results of justice-based footprints in specific nations, we select the top 20 countries based on the justice-based footprints for plant losses and vertebrate losses, respectively. The top 20 countries with the largest environmentally just responsibility contribute 71% of plant losses and 68% of vertebrate losses separately (Fig. 3). In some countries more than 90% of environmentally just responsibility of plant losses are driven by domestic consumption, for example in Haiti (96%), South Africa (94%), Madagascar (92%), and India (92%). In contrast, environmentally just responsibility of plant losses in some nations, such as Japan (61%) and the US (50%), mainly derives from consumption of imported products, as consumption in these nations heavily relies on imports. Exported products in some nations, such as Australia (43%) and Sri Lanka (36%), account for a large share of environmentally just responsibility for plant losses. Environmentally just responsibility for vertebrate losses sees a similar pattern to plant losses, while domestic consumption contributes slightly less than that in plant losses. For instance, domestic consumption drives the largest share of vertebrate losses in Madagascar (90%). Consumption of imported products dominates the environmentally just responsibility of vertebrate losses in Japan (80%).

Trade-related biodiversity loss is the actual part of shared responsibility in the environmentally just framework. As such, an justice-based footprint with a higher proportion of trade-related biodiversity loss may need more attention. To zoom in justice-based footprints with higher proportions of trade-related biodiversity loss in individual nations, we compare the environmentally just accounting to the average of production- and consumption-based accounting (hereafter as relative environmentally just accounting) for the top 20 countries ranking by the absolute value of relative justice-based plant losses and vertebrate losses (Fig. 4). For example, both Niger and Norway heavily relied on imported commodities from the international market, with imported commodities contributing 91% and 87% to Niger's and Norway's justice-based plant losses, respectively. However, they see different directions for relative environmentally just accounting (Fig. 4A). Norway sees more responsibility, while Niger sees less responsibility, since Niger has a low HDI at 0.283 compared to Norway's 0.932. Similarly, in terms of vertebrate losses, Solomon Islands and Ireland, both with a high proportion of trade-related biodiversity loss, see a different direction for relative environmentally just accounting due to the difference between HDIs (0.487 in the Solomon Islands and 0.896 in Ireland). The Solomon Islands see a higher proportion of exports which accounts for 66% of the justice-based footprint, while Ireland has a higher proportion of imports which contributes 92% to justice-based vertebrate losses.

4. Discussion

Consumers and producers both share responsibility for environmental damages such as biodiversity loss. Accounting for this responsibility will become increasingly important in international trade policy as efforts to improve biodiversity protection hopefully increase. In the international market, countries in the Global North benefit most (Fig. 3), as they tend to outsource land use and its associated biodiversity loss to the Global South with lower regulatory standards and higher biodiversity. For example, consumption- and production-based plant losses from the land use of the Global North are 1300 and 920 species respectively, with 3739 and 4119 species in the Global South. This implies the Global North sees a net import of 380 plant species lost in the Global South. Economic growth will likely threaten biodiversity loss further, especially in rapidly growing regions (Marques et al., 2019). For example, the share of biodiversity loss transferred through international trade dropped from 69% to 48% in Western Europe and North America between 2000 and 2011, while the share increased from 13% to 23% in Asia and the Pacific (Marques et al., 2019). Increasing globalization urges to consider shared responsibilities of biodiversity loss among all agents along global value chains.

Biodiversity is considered a public good, and its conservation requires significant improvements in governmental and international policy (Rands et al., 2010). International trade presents a specific challenge, since complex supply chains and regulatory differences result in biodiversity-loss leakage (analogous to carbon leakage) whereby environmental damage is exported to overseas producers. To abate carbon leakage, the EU is looking to implement a carbon border adjustment mechanism by the end of 2021 (European Commission, 2019). However, the EU will only aim to identify the main drivers of biodiversity loss from 2021 onwards and is yet to take any overarching action to directly prevent biodiversity decline along trade networks (European Commission, 2019). Adding biodiversity conservation costs onto the price of goods, or levying a tariff to protect biodiversity are potential ways to allocate biodiversity conservation responsibility while limiting leakage (Wilman, 2019).

The financial support for biodiversity protection efforts is not sufficient. For example, $150–440 billion per year would have been needed to meet 20 Aichi Biodiversity Targets by 2020 (CBD, 2014), while the expenditure was less than half at $78–91 billion per year, with most ($67.8 billion per year) derived from domestic public expenditure (OECD, 2020). Intergovernmental cooperation to develop appropriate policy interventions for biodiversity conservation is still at a very early stage. For example, biodiversity has attracted international expenditure of only $3.9–9.3 billion per year (2015–2017 average) (OECD, 2020). In comparison, a single international climate change agreement attracted $100 billion in pledges from developed countries for developing countries by 2020 (Barbier et al., 2018). Indeed, ongoing aims are to raise $100 billion a year until 2025 (even though both the 2020 and 2025 pledges are yet to be met). The CBD under which the Aichi targets fall generally follows a combination of contributor pays and ability to pay principles (Lehmann, 2017), similar to the justice principles considered here. While these principles assign developed countries key responsibilities to fund conservation in developing countries, recent decisions at Conferences of the Parties (COPs) diverge from this by attempting to shift responsibility to developing countries through increased domestic funding and South-South cooperation (Lehmann, 2017). The environmentally just accounting identified national responsibility for global biodiversity loss. As such, the results can provide a guide for national actions and international cooperation of biodiversity conversations.

It is important to provide an integrated view not only across nations but also sectors. The food system is the largest driver of global terrestrial biodiversity loss (IPBES, 2019; Rockström et al., 2020). Furthermore, the provision of food, feed, and other materials for human needs has been increasing at the expense of other nature's contribution to people

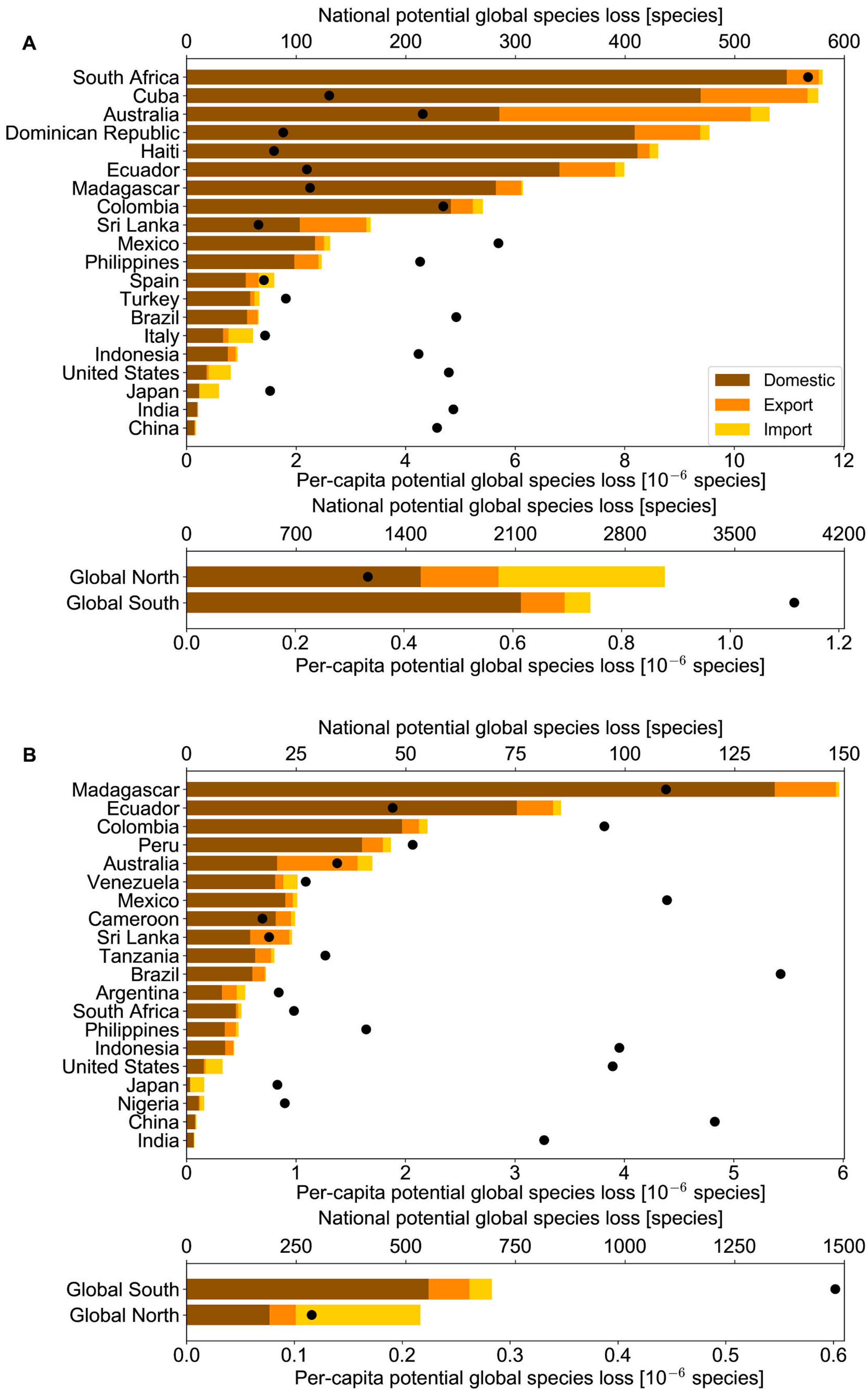

Fig. 3. Environmentally just responsibility for biodiversity loss given by domestic consumption, export and import for the top 20 countries, the Global North and Global South for A) plants, and B) vertebrates (mammals, birds, amphibians, and reptiles). The bars represent per-capita values, while the dots represent national values.

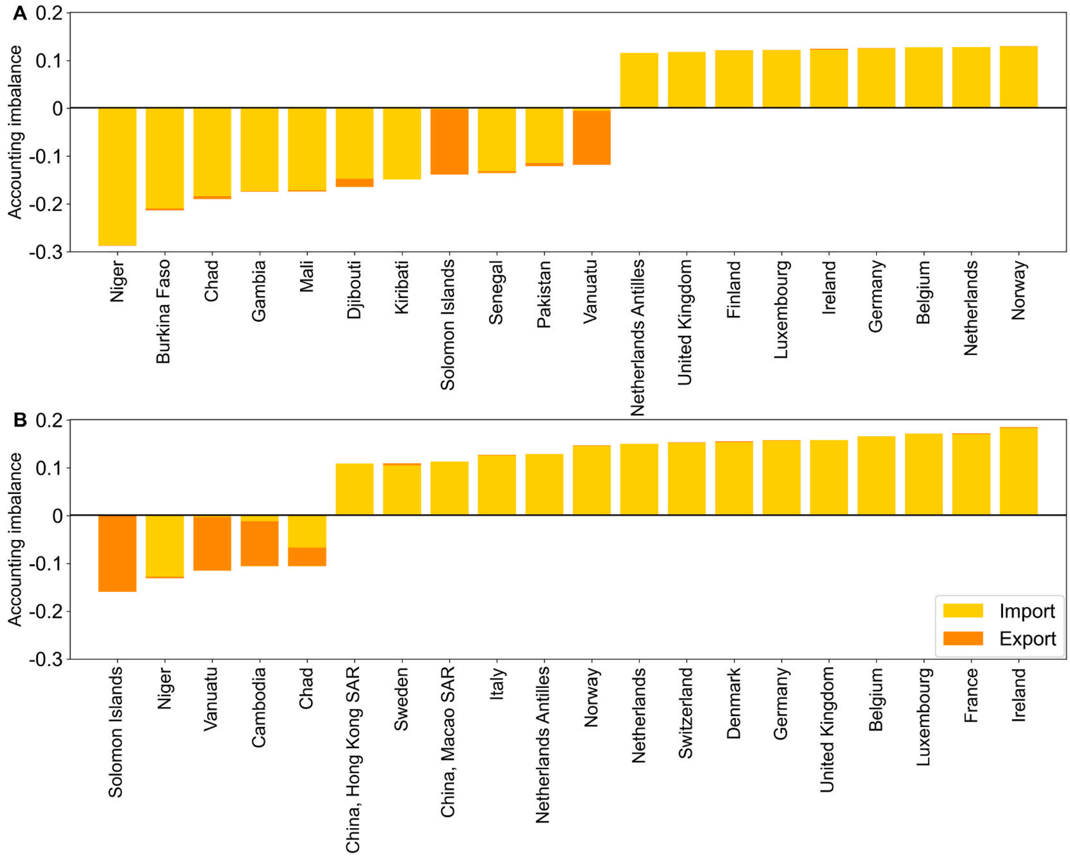

Fig. 4. Justice-based accounting in relation to the average of production- and consumption-based accounting for plants (A) and vertebrates (B) for the top 20 countries. An accounting imbalance implies a gap between justice-based accounting based on the HDI and the average of production- and consumption-based accounting.

(Díaz et al., 2019; IPBES, 2019). In terms of global supply chains, farmers who rely on agricultural production for their livelihoods are only able to acquire a relatively small portion of consumer expenditure on foods (Yi et al., 2021). As such, environmental justice should be considered in responsibility sharing between producers and consumers of the food system along global supply chains. To abate biodiversity decline, policies on sustainable food production and consumption are particularly important and can greatly impact land use and, therefore, biodiversity (Crist et al., 2017; Delabre et al., 2021; Henry et al., 2019). Dietary change, reduction of food waste, and technologies for increasing yields are all policies, which put both direct and indirect pressures on global biodiversity and could be coordinated transnationally to target specific regions of biodiversity loss (Delabre et al., 2021; Leclère et al., 2020; Willett et al., 2019).

Although the HDI has been arguably the most successful multidimensional indicator in the past three decades, the index has received several critiques on the basis of variables used and its computational method (Deb, 2015; Herrero et al., 2012). In terms of variables, Herrero et al. (2012) recommended using 'expected years of schooling' rather than both 'mean years of education' (for adults) and 'expected years of schooling' (for children) to remain consistent with the life expectancy index and transparent in the calculation (Herrero et al., 2012). Some studies have criticized the fact that the HDI covers only three dimensions (i.e. health, education, and income) and misses others, such as economic and social cohesion or human rights dimensions (Deb, 2015). Other critiques focus on the computational method of HDI. The updated HDI in 2010 employs the geometric mean of normalized indices, which reduces the substitutability and is believed to generate more sensible rankings (Zambrano, 2017). However, critiques centered on the use of the same weight to the three dimensional indices remain (Deb, 2015; Herrero et al., 2012). These critiques about the HDI may affect the justice-based footprint. The HDI as a justice parameter is an attempt to operationalize the justice-based footprint framework across the global economy. Covering the various dimensions of justice more directly is challenging due to limited data availability and requires further research (Oliveira, 2019). Future research could use alternative justice weighting parameters but could also conduct an analysis based on more recent data, considering increasing trade volumes, further land use change, and land use intensity changes. For instance, land use change since 2005 may cause further biodiversity loss. From 2005 to 2019, pasture decreased by 2.4%, while cropland and urban area increased by 1.3% and 8.3% (Winkler et al., 2021).

5. Conclusion

As an urgent, global, complex problem, biodiversity loss represents a critical common action problem similar to other global environmental problems such as climate change. Building a better picture of this global loss and how the responsibility of this loss is shared across nations is important for further integration of science, policy, and justice in international frameworks. The environmentally just accounting based on integrated FABIO and EXIOBASE datasets provides a new perspective to allocate the responsibility for environmental footprints, here biodiversity loss, between producers and consumers. We employ the HDI as a weighting factor to approximate environmentally just responsibilities between producers and consumers. The results show that the Global North shares more responsibility under environmentally just accounting than the simple average of production- and consumption-based accounting. From an environmentally just responsibility perspective, we can see most countries in the Global North shoulder more responsibility from import and domestic consumption (except for biodiverse countries such as Australia), while most countries in the Global South shoulder more responsibility from domestic consumption and export. We hope this work constitutes a further step towards both understanding biodiversity loss and providing a basis for cooperation in implementing solutions for the post-2020 global biodiversity targets.

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.

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Appendix A. Supplementary data

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