Article:

Molotoks, Amy, Green, Jonathan Michael Halsey orcid.org/0000-0002-5003-0203, Ribeiro, Vivian et al. (2 more authors) (2023) Assessing the value of biodiversity-specific footprinting metrics linked to South American soy trade. People and Nature. ISSN 2575-8314

https://doi.org/10.1002/pan3.10457

Received: 30 April 2022 | Accepted: 14 October 2022

DOI: 10.1002/pan3.10457

PEOPLE AND NATURE | BRITISH ECOLOGICAL SOCIETY

RESEARCH ARTICLE

Understanding Land-Use Driven Biodiversity Change

Assessing the value of biodiversity-specific footprinting metrics linked to South American soy trade

Amy Molotoks[1] | Jonathan Green[1] | Vivian Ribeiro[2] | Yunxia Wang[3] | Chris West[1]

[1]Department of Environment and Geography, Stockholm Environment Institute York, University of York, York, UK

[2]Stockholm Environment Institute, Stockholm, Sweden

[3]Genetics and Conservation, Royal Botanic Garden Edinburgh, Edinburgh, UK

Correspondence
Amy Molotoks
Email: amy.molotoks@york.ac.uk

Funding information
Global Challenges Research Fund, Grant/Award Number: ES/S008160/1; Gordon and Betty Moore Foundation, Grant/Award Number: 7703.01; Norway's International Climate and Forest Initiative, Grant/Award Number: QZA-21/0156; Quadrature Climate Foundation, Grant/Award Number: 01-21-000098

Handling Editor: Laura Graham

Abstract

1. International demand for a small handful of commodities is a major driver of tropical deforestation and associated biodiversity loss. Previous commitments to reduce commodity-driven deforestation have largely failed, yet there are currently various proposals in place globally, which aim to address the challenge of reducing overseas environmental impacts of supply chains.
2. However, many of these are highly focused on the issue of deforestation alone. Given that biodiversity objectives are often cited alongside protection of forests, deforestation rates are therefore often used as a proxy for biodiversity loss. Assessments exploring deforestation risk linked to commodity supply chains, enabled by increasingly granular information on sourcing patterns, therefore potentially overlook other important biodiversity concerns.
3. In response, we examine sourcing risks across three producer countries in South America for both forest loss and biodiversity for the example of soy production and trade, which has one of the largest embodied deforestation footprints in international supply chains. Using IUCN and Birdlife data, we create four simple biodiversity metrics to represent different aspects of species-related biodiversity risk and link both these, and a forest loss metric representing soy driven deforestation, to sub-national supply chain data to examine risks for the two largest importers from each producer country.
4. We find relatively little evidence of convergence between forest loss and biodiversity metrics, as well as divergence between the four biodiversity indicators both for different importers and across landscapes. This suggests not only that forest loss alone is unlikely to be an adequate proxy for biodiversity, especially at larger spatial scales when considering risks across sourcing patterns, but also that further work is necessary to develop a deeper understanding of interactions between more complex measures of biodiversity and their consequences for informing supply chain activities.

KEYWORDS
biodiversity, South America, soy, supply chains, trade

[Correction added on 28-02-2023, after first online publication: The author name has been corrected from "Vivian Ribiero" to "Vivian Ribeiro"].

1 | INTRODUCTION

A series of voluntary corporate commitments to remove deforestation from supply chains, made in the early 2010s, have yet to make significant progress in slowing or reversing rates of forest loss (Garrett et al., 2019; Garrett & Rueda, 2018; Smit et al., 2020; Zu Ermgassen et al., 2020). There are now several proposals for further, regulatory, measures, including legality-based due diligence for forest risk commodities in the UK (DEFRA, 2021); the proposed regulation to minimise EU-driven deforestation and forest degradation in the EU (European Commission, 2019, 2021); and the Fostering Overseas Rule of law and Environmentally Sound Trade (FOREST) Act of 2021 in the US (GPO, 2021). The introduction of such legislation provides both opportunities and risks for biodiversity conservation. It is important to understand the interplay between actions to protect forest habitats and broader considerations for species conservation, which is often the primary target for conservation initiatives, explicitly addressing both the potential for co-benefits or trade-offs. However, the extent to which zero deforestation commitments and regulation can tackle commodity-driven biodiversity declines in terms of risk to species loss is relatively unexplored.

The increasing demand for agricultural commodities is a major driver of global deforestation and there has been little progress made in reversing these trends (Curtis et al., 2018). This is a particular concern in the biodiverse tropics where species and ecosystems are threatened by the expansion of commodity production (Pendrill et al., 2019). The globalisation of trade means that consumers and producers are connected across large distances and tropical deforestation can be driven by consumption elsewhere (Defries et al., 2010). A large proportion of tropical deforestation and associated biodiversity loss can, however, be attributed to international demand for just a small number of commodities (Henders et al., 2015). Globally, 80% of all threatened terrestrial mammal and bird species are affected by agriculturally driven habitat loss (Tilman et al., 2017) and habitat loss is the main direct cause of biodiversity declines (Joppa et al., 2016).

International demand for soy, which is primarily used as feed for livestock, with only 6% being directly for human consumption (WWF, 2014), has one of the largest embodied tropical deforestation footprints of any agricultural commodity (Henders et al., 2015). This 'footprint' refers to the area of deforestation attributed to production of a commodity, and within a supply chain context, 'embodied' refers to impacts from production of a commodity that is exported, and hence, the impacts in producer regions are attributed to the importer country which is driving the demand. Soy has also historically been a key driver of the conversion of intact, tropical forests to cropland across large parts of South America (Gibbs et al., 2010). Together with palm oil, soy accounts for over a fifth of total embodied tropical and subtropical deforestation in international trade (Pendrill et al., 2019). Brazil, Argentina and Paraguay produce over 50% of the world's soy (FAO, 2022) and international demand continues to drive further deforestation and loss of natural vegetation, most notably in the Cerrado, a tropical savannah biodiversity hotspot (Strassburg et al., 2017) despite commitments to reduce embodied deforestation within supply chains (Gibbs et al., 2015).

Export of forest-risk commodities is also, however, important for national development objectives and the benefits of agricultural production, such as food security (Delzeit et al., 2017) and economic profits (Naidoo & Iwamura, 2007) may compete with biodiversity conservation goals. This is particularly the case in tropical countries where agricultural production tends to make a larger contribution to countries' GDP (World Bank, 2022) and there are direct positive social impacts from trade including income, nutrition and living standards (Dreoni et al., 2022). Yet humans depend upon biodiversity, and large-scale losses induced by expansion of agriculture will impair ecosystem function and services, with direct consequences for livelihoods, and hinder progress towards multiple sustainable development goals (Dasgupta, 2021; IPBES, 2019). Despite this, the majority of corporate target setting and legislative attention is focused on the conversion of ecosystems, particularly forest (Marshall et al., 2019). However, the addition of biodiversity information offers the chance to both distinguish areas of greater biodiversity concern within forests as biodiversity value differs from forest to forest (Condit et al., 2002; Gardner et al., 2009) and identify other habitats that contribute to biodiversity conservation goals.

There are a multitude of indicators that represent different components, or aspects, of biodiversity, or present it in different ways (Ferrier, 2002). Although some studies comparing consumption-based biodiversity footprints have shown some convergence of such indicators on a global scale, it is limited for others, demonstrating the importance of including a range of biodiversity metrics in footprinting assessments (Davies & Cadotte, 2011; Marquardt et al., 2019). Furthermore, congruence is highly scale dependent; for example, at fine spatial resolutions, hotspots of rarity and threat across taxonomic groups may not overlap (Grenyer et al., 2006). This demonstrates the relevance of including a range of biodiversity metrics in impact assessments. Marques et al. (2021) suggests that assessments should, at a minimum, represent two complementary aspects of biodiversity for example a measure of extinction risk of species alongside one representing ecosystem function. When considering interactions between biodiversity risk and commodity-trade systems, there are studies on the impacts of trade on species threat at national scales (e.g. Lenzen et al., 2012) and those that examine agricultural production impacts on biodiversity at high spatial resolution (e.g. De Baan et al., 2015). Yet to our knowledge, there is just one study that incorporates both the impact of production and trade of an agricultural commodity on species threat at high, subnational spatial resolution (Green et al., 2019). There are also none that compare different species related measures of biodiversity risk with one focusing on a loss of a specific ecosystem, that is a deforestation assessment, and discuss the implications for trade-linked risk assessment.

In response, through use of trade data from the Trase Spatially Explicit Information on Production to Consumption Systems (SEI-PCS) model of subnational production and export (Godar et al., 2015; Trase, 2022), this study examines soy supply chains for the three largest exporters in South America: Argentina, Brazil and Paraguay (FAO, 2022; Trase, 2022). The use of this high-resolution data on production and trade allows us to consider land use change driven by the soy consumption patterns of consuming regions and to compare a metric of deforestation risk against a series of species related biodiversity risk measures. This analysis, for example, helps elucidate potential gaps, for biodiversity, in any protection offered by the actors involved in downstream supply chains; for example, those linked to due diligence legislation that focuses on deforestation risks. Conversely, it can help to identify any threats that may emerge following a redistribution of sourcing away from areas of commodity-driven deforestation. In other words, we ask: does the application of biodiversity risk indicators as an alternative to a metric of deforestation risk change conclusions, from the perspective of stakeholders sourcing soy, on their key areas of environmental concern? Or instead, does a deforestation indicator act as an adequate proxy for these concerns that can encompass other important aspects of biodiversity such as species vulnerability and rarity?

2 | METHODS

A range of open source data was used to calculate and run the metrics for assessing the potential biodiversity or forest loss risk from soy production and trade.

2.1 | Land use data

Data representing the presence of soy production from the Global Land Analysis and Discovery (GLAD) laboratory, University of Maryland was used from 2001 to 2019 at a resolution of 30 m (Song et al., 2021). Original files were filtered by areas of soy with at least 20 ha to ensure other land uses were not mistaken for soy, and a multiband raster for South America produced with each band representing the increment of soy for each given year. For our analysis, the years 2001–2018 were selected to produce a dataset of the total coverage of soy production over this time period, as the latest trade data of soy from Latin America from Trase is only available for 2018.

Global forest change data, also from the University of Maryland, was used to represent forest loss (Hansen et al., 2013). The time series analysis of Landsat images characterises global forest extent and change from 2000 to 2020. Version 1.8 of this dataset was used and the time period 2000–2017 was selected for analysis of forest loss as a result of soy expansion, using the data layer 'Forest Loss'. This allows for a lag period of 1 year between a deforestation event and the first possible harvest of soy. The multiband raster contained a layer of global forest cover loss, defined as stand-replacement disturbance, or a change from a forest to a nonforest state. This layer was selected and then used to calculate the forest loss attributed to production of the chosen commodity. Given the data source used, we refer to this metric as forest loss, however as it constitutes the complete removal of trees for the permanent conversion of forest to another land use, it is used in this study to represent commodity driven deforestation.

2.2 | Biodiversity

Data for the global mapped ranges of 11,145 bird species (BirdLife, 2021), 6707 amphibians, 5537 terrestrial mammals and 10,148 reptiles (IUCN, 2021) were downloaded from IUCN and BirdLife portals. These species range polygons, also known as 'extent of occurrence', contain the smallest area encompassing the known, inferred or projected sites of present occurrence of a species. They are not refined by habitat, altitude or biotic interaction and therefore are likely to contain large areas which are unsuitable for species, hence the data only represents the potential biodiversity in the area. Yet it also represents a precautionary measure of the degree to which species might interact with habitats and land use types.

The spatial data also contains information on extinction risk, presence, origin and seasonality. For all biodiversity metrics analysed, we filter the original data according to specific criteria. All extant species (i.e. presence is 'Extant' and they are not listed as 'Extinct' or 'Extinct in the Wild') and all parts of seasonal ranges were included. For origin, species that are either native to the area or originate from reintroductions or assisted colonisation (i.e. conservation interventions) were included. These filters were applied to all metrics generated from these data.

2.3 | Metrics

To create four measures of biodiversity, data downloaded from the IUCN and Birdlife portals was used to create (a) total, (b) threatened, (c) endemic and (d) forest habitat affiliated species risk metrics (Figure 1). Although these are all species related metrics, they cover different aspects of biodiversity which may be considered important when approaching species conservation, that is, vulnerability, endemism and habitat dependency. The first step in creating these metrics was to create a species richness layer, including the following taxonomic groups; mammals, birds, amphibians and reptiles. This analysis was conducted in Google Earth Engine by counting all species potentially present on a 1 km resolution. This created a total species richness layer representing all species present, whereas the following three metrics were created using subsets of these data.

The subsets of data were created using the following criteria. The extinction risk categories endangered, critically endangered and vulnerable species were selected for threatened species. A list of endemic species (defined as species occurring naturally within only one country) were downloaded separately from the IUCN portal for endemic species. A 'forest habitat affiliated (FHA) species' metric was defined by species' habitat affiliations: selecting the IUCN Habitat Classification of 'forest' assumed all species with that habitat affiliation were forest dependent. For each of these three subsets of

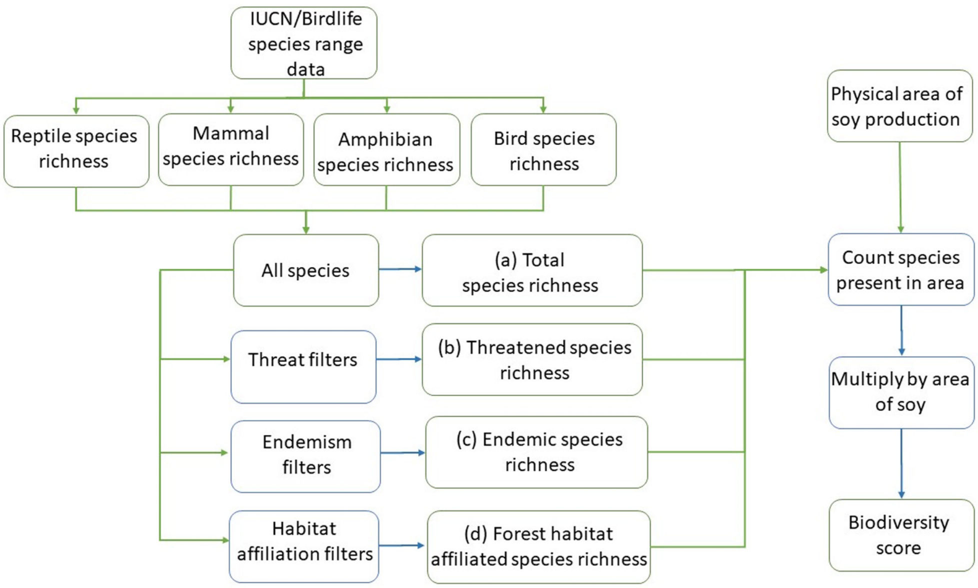

FIGURE 1 Flow chart showing the methods for creating biodiversity score for each biodiversity metric.

data, a species richness layer composed of the same four taxonomic groups was also created.

All four species richness layers, total, threatened, endemic and FHA, were then intersected individually with the total soy coverage map. A biodiversity score was calculated by multiplying the species count by the physical area of cropland in each pixel to represent the risk to biodiversity from soy production in each pixel. Hence, if there were 10 species present and 14 ha of cropland for soy production within a pixel, the biodiversity score would be 140. This therefore represents a measure of potential risk to biodiversity loss from commodity production. Similarly, for the forest loss metric, the area of forest loss intersecting with the physical area of cropland for soy production was calculated for each pixel to attain a deforestation score. Both this and the four biodiversity score results for total, FHA, endemic and threatened species were each summed within a grid of 10 × 10 km across the three producer countries to show the spatial variability between the metrics before being linked to the trade data (Figure 2).

2.4 | Trade data

Material flow data for 2018 from the trase platform were downloaded (Trase, 2022), including volume of soy produced and the municipality or department identification code for Brazil, Argentina and Paraguay. All trading companies and importers were selected. Once downloaded, the unique identification code was used to match trade data with metric results from each region; either municipality or department depending on the producing country. For each metric, the biodiversity score was divided by the total production of soy in tonnes for the subnational region to get an intensity score. This was then multiplied by the percentage of trade in total soy that was traded by the importer from each producer country to derive the final risk score (Figure 3).

Scores were also summed per municipality or department within the top two importer trade flows; the European Union across all three producer countries, as well as China (which refers specifically to mainland China only throughout) for Argentina and Brazil, and Argentina for Paraguay. Argentina was chosen as the comparator country for Paraguay sourcing as China did not import soy directly from Paraguay in 2018 and Argentina is the second largest importer. For each producer country, the proportion of the risk scores within the two trade flows for each municipality or department was calculated and scores ranked from highest to lowest for each metric. A cumulative sum was then calculated to show how many municipalities or departments each metric requires to reach arbitrary, indicative biodiversity risk thresholds and to demonstrate differences across the different metrics. For each metric, the risk scores for each importer were also divided by the total risk across all trade flows to allow comparison between importers of relative contribution to total risk.

3 | RESULTS

The biodiversity score results demonstrate the spatial variability of the different metrics, highlighting different hotspots where there is a higher risk to biodiversity associated with soy production (Figure 2). Using Argentina as an exemplar, results for the European Union and China are shown (Figure 4). These two economic blocks are the two largest importers of soy produced in Argentina, importing 6.24 and 3.99 million tonnes of soy in 2018, which is approximately 22% and 14% respectively of the total soy exported (Trase, 2022).

Figure 4 shows the percentage of the cumulative risk score within a certain number of departments for each metric; that is, it shows how much of the total risk for each metric is encompassed within a specific number of departments, or how many departments it takes

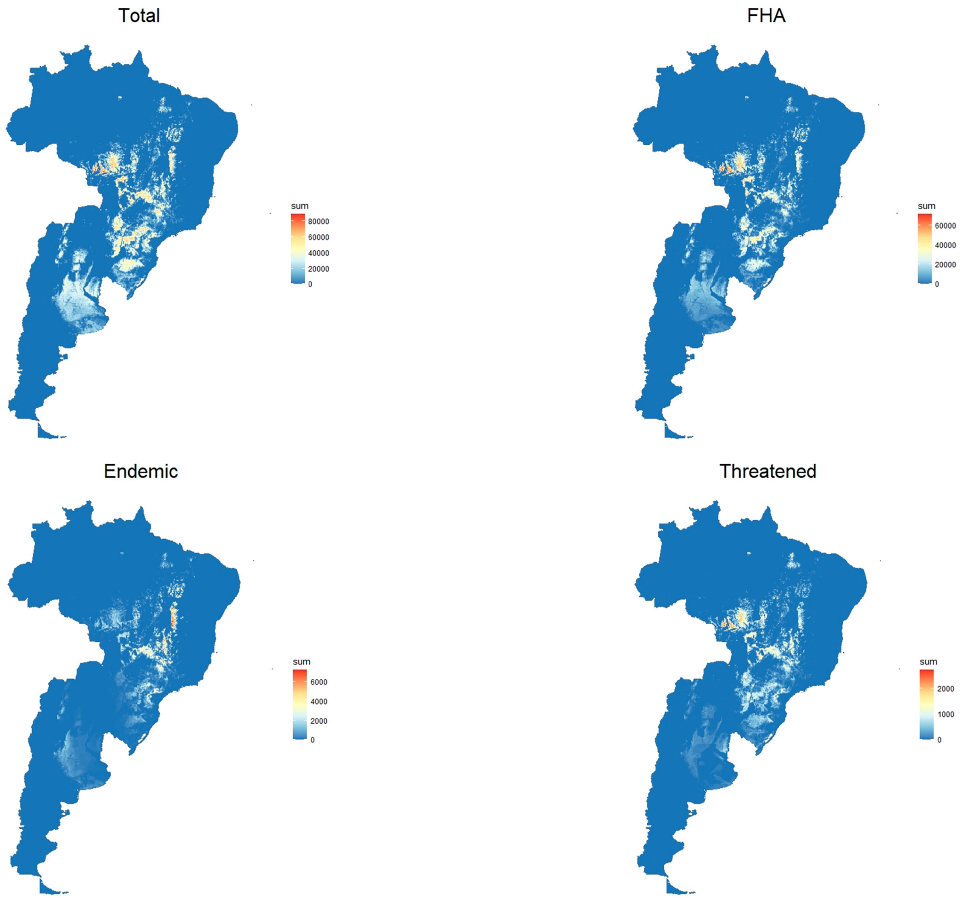

FIGURE 2 Spatial variability of metric scores across 10 × 10 km grid of three countries Argentina, Brazil and Paraguay.

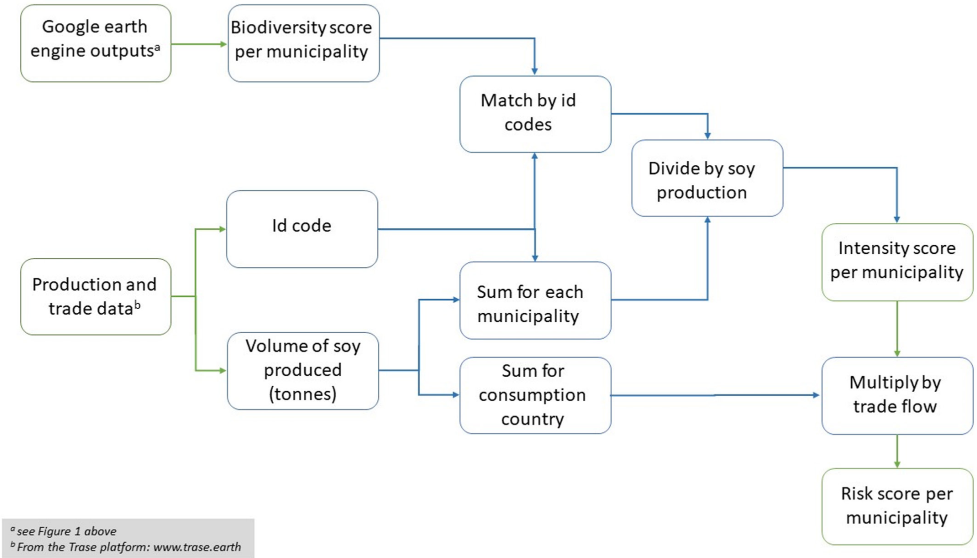

FIGURE 3 Flow chart showing the methods for linking biodiversity metric scores to trade data.

to reach a certain 'threshold' or percentage of the total risk score. When ranked according to the cumulative sum for forest loss, the biodiversity metric scores consistently require more departments for both importers to reach biodiversity risk thresholds as shown by the steeper gradient of the line for forest loss (Figure 4). Therefore, if an importer was interested in understanding which regions of production linked to their trade could mitigate a certain percentage of forest loss risk e.g. if production processes were improved with sustainability objectives a significantly lower percentage of other biodiversity risks would be covered.

For example, using an arbitrary threshold of 80% of their total forest loss risk exposure, Figure 4 shows that only the equivalent of 15% and 13% of total species risk for EU and Chinese trade respectively would also be covered in the same number of departments (Figure 4, Table 1). This suggests that, here, forest loss is an inadequate proxy for other species-related aspects of biodiversity risk, which can also been seen if another risk threshold was chosen for the cumulative risk score. Forest loss risk is concentrated in far fewer departments for both importers compared to the interaction of their sourcing patterns with other aspects of biodiversity (Figure 4, Table 2).

This is also the case for Brazil which shows the same pattern with both importers, albeit with less pronounced divergence between the forest and biodiversity metrics than for Argentinian soy trade (Tables 1 and 2). For example, an 80% forest loss risk threshold would incorporate a larger percentage of other biodiversity risk metrics, particularly for EU trade, although the highest still only reaching 52% for threatened species (Table 1). It is also worth noting that for both Argentina and Brazil, it takes substantially fewer departments or municipalities for EU trade to reach an 80% risk threshold compared to China, hence a larger amount of both forest loss and biodiversity risk is concentrated in a smaller area for the EU supply chain (Table 2).

The pattern of biodiversity metrics taking more departments and land area to reach an 80% threshold is duplicated for EU trade of soy produced in Paraguay (Table 2). However, although forest loss still diverges from biodiversity metrics, the opposite pattern is shown for exports to Argentina. Here, forest loss requires more departments compared to other metrics to reach an 80% threshold (Table 2). This suggests sourcing patterns have less risk associated with forest loss than other biodiversity metrics shown here compared to EU trade. It is worth noting patterns are less easy to interpret for Paraguay due to the difference in spatial scale of the analysis for this producer, with data only being available for a larger administrative division. Departments in Paraguay are a different administrative level to departments in Argentina or municipalities in Brazil. Hence, we have fewer results for Paraguay and at a coarser spatial scale than the other two producer country cases. This is reflected in the number of departments required to reach an 80% risk threshold only differing by one across forest and biodiversity metrics (Table 2) and the results showing forest loss to be more aligned with biodiversity metrics (Table 1). Therefore, the extent to which forest loss is shown to be a good proxy for biodiversity metrics could be scale dependent.

As well as divergence between forest loss risk and biodiversity metrics across landscapes and importers, there is also variation between biodiversity metrics. For example for Argentina, trade with China shows fairly good congruence between a few metrics up to a point, then endemic species risk diverges and becomes more aligned with threatened, which diverges much sooner (Figure 4b). Almost the opposite is shown for EU trade, with most biodiversity metrics converging at a higher threshold, however threatened species risk still shows less congruence with other metrics for this importer (Figure 4a).

This variation and mixed levels of congruence are also seen for the other producer countries, for example for EU trade with Brazil, there is convergence between endemic and total species risk, but endemic diverges to become more aligned with others at higher

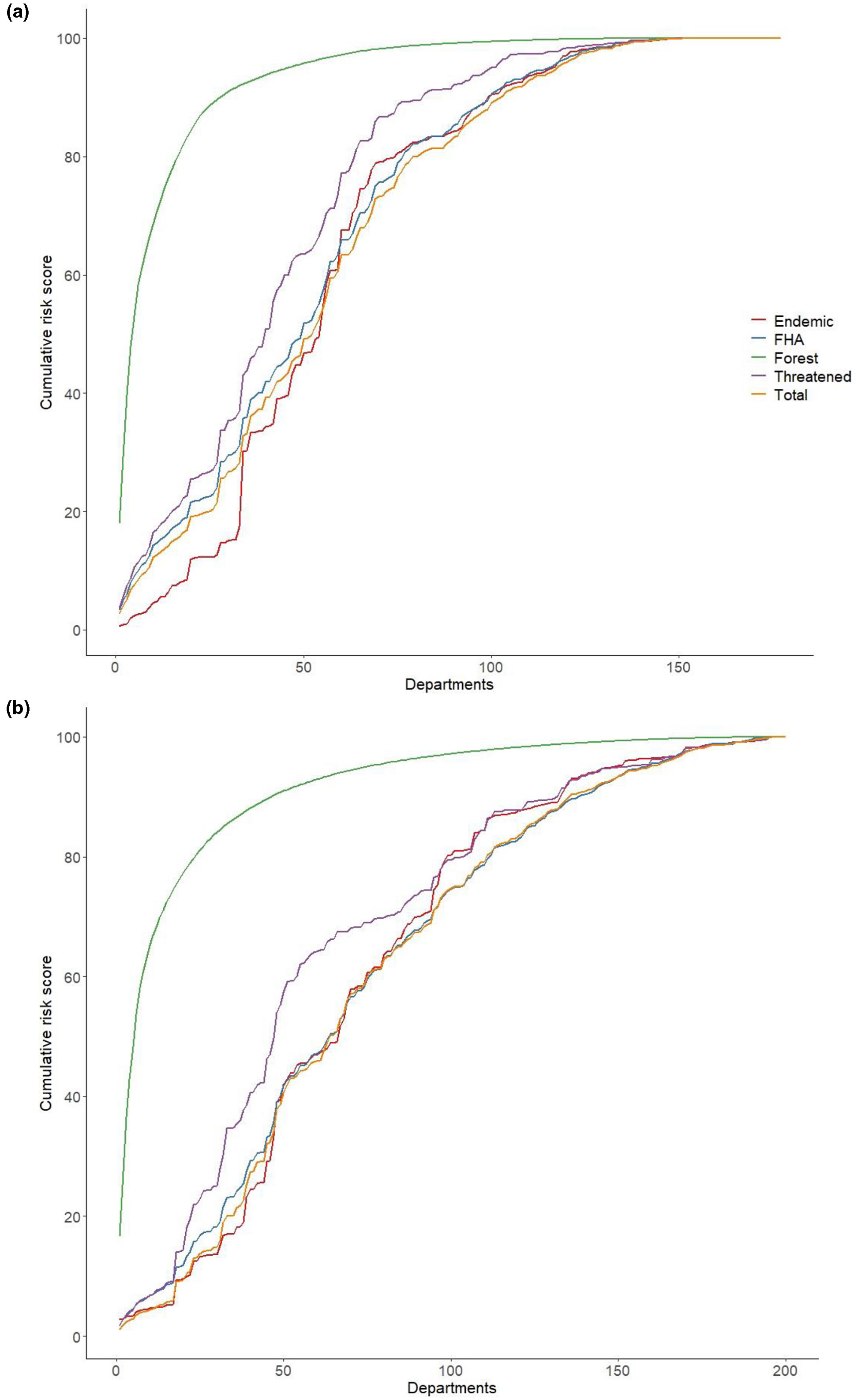

FIGURE 4 Cumulative ranked risk scores by forest loss risk (in green) for (a) EU and (b) Chinese trade of Argentinian soy.

risk thresholds (Figure S3). For Chinese trade, there is less convergence between metrics, and less variation in the patterns shown (Figure S4). EU trade with Paraguay also shows more congruence between metrics than for trade with Argentina (Figure S5). The main conclusion when contrasting different metrics of biodiversity is therefore that the area required, that is, number of administrative units, for thresholds to be reached and the extent to which there is congruence between different metrics is not consistent. This

TABLE 1 Percentage of biodiversity risk score encapsulated within departments or municipalities that account for 80% forest loss from soy production for each producer country.

	Argentina		Brazil		Paraguay	
Risk metric	EU	China	EU	China	EU	Argentina
Threatened	21%	22%	52%	35%	74%	91%
FHA	17%	15%	51%	35%	75%	92%
Endemic	8%	12%	44%	40%	75%	94%
Total	15%	13%	45%	30%	75%	91%

TABLE 2 Number of departments or municipalities needed to exceed 80% risk threshold for each metric and each producer country.

	Argentina		Brazil		Paraguay	
Risk metric	EU	China	EU	China	EU	Argentina
Forest loss	17	24	35	62	4	5
Threatened	64	104	139	420	5	4
FHA	77	112	155	456	5	4
Endemic	75	99	145	372	5	4
Total	81	111	176	509	5	4

inconsistency is also apparent when we consider different producer countries and importers.

The results shown in Figure 4 for the cumulative risk scores can also be shown spatially, demonstrating where the departments with the highest percentage of total risk are. Given that the cumulative risk score is ranked from largest to smallest, Figure 5 therefore shows the minimum number of departments needed to reach a percentage of the total risk score for each metric and importer. Spatial representation of risk scores further demonstrates forest loss risk being concentrated in fewer departments and different locations compared to biodiversity metrics.

For example, in Figure 5, 20% of forest loss risk, that is, areas in red, for both the EU and China is from only two departments in Argentina. However, the number is greater for both importers for endemic species risk, with EU and Chinese trade requiring three and five departments respectively to encapsulate 20% of total risk (Figure 5c,d). Furthermore, there is greater spatial congruence of forest loss risk between importers than there is for other metrics. Both EU and Chinese trade show the greatest forest loss risk in the north of Argentina, whereas for endemic species, the departments with the highest risk are not shared across importers (Figure 5). This is also the case for other metrics (Figures S1 and S2).

These patterns are also seen for Brazilian trade, with forest loss being more highly concentrated than other metrics in similar locations across importers (Figures S7 and S8), particularly for China which has a much more widely distributed risk compared to the EU which is concentrated in fewer municipalities for all metrics (Table 2). There are also differences between biodiversity metrics within importer supply chains across space. For example, FHA and endemic species appear well aligned for Chinese trade with Argentina (Figure S2) yet are drastically different for Chinese trade with Brazil. For Brazil, endemic species risk is more highly concentrated further north east whereas FHA species risk is more scattered across municipalities further east (Figure S8).

These spatial differences are not as apparent for Paraguay due to the number of departments being considerably fewer and on average, larger (Figures S9 and S10). However, there is still notable divergence, particularly for EU trade between forest loss and biodiversity metrics for EU trade with Paraguay (Figure S9). Here, the same department, Alto Parana, contributes to 20% of the total risk for all metrics apart from forest loss for which the department with the largest individual contribution to risk, San Pedro, is located further west (Figure S9).

The risk scores for each metric were summed for all trade with the two largest importers for each producer country. These were normalised by dividing each importers total risk score by the total risk score across all trade flows from the regions of production. Hence, the results show the proportion of risk for each trade flow of the total risk score if all importers were considered. These risk scores are shown (Figure 6) for each metric to demonstrate the relative risk associated with each importers consumption of soy and to enable direct comparison between them.

For Argentina, the normalised risk scores for trade with the EU and China are similar in terms of their total contribution to overall risk, with the most similar being for total species risk (Figure 6), however; there is variation across the different metrics, with greater variance for trade with China. For example there is a much larger percentage difference between the metric with the highest (endemic) and lowest (forest loss) impact values (Figure 6). Chinese trade also has a higher overall risk on endemic species compared to the EU, despite importing less soy, which suggests risk on endemic species is disproportionately associated with trade with China. The EU however has a substantially larger impact on forest loss compared to China (Figure 6), with the highest score being for forest habitat affiliated species.

As with the department or municipality level scores, patterns vary both with producer country and across metrics between importers. For example, EU trade with Brazil and Paraguay shows the highest total risks to forest loss (Figures S11 and S12), whereas this is the metric with the lowest score for trade with Argentina (Figure 6). The extent to which risk scores vary also differs, for example, the overall risk scores are more similar for China when trading with Brazil (Figure S11) compared to Argentina (Figure 6). Therefore, both the pattern and the scale of variation differ across both producers and importers.

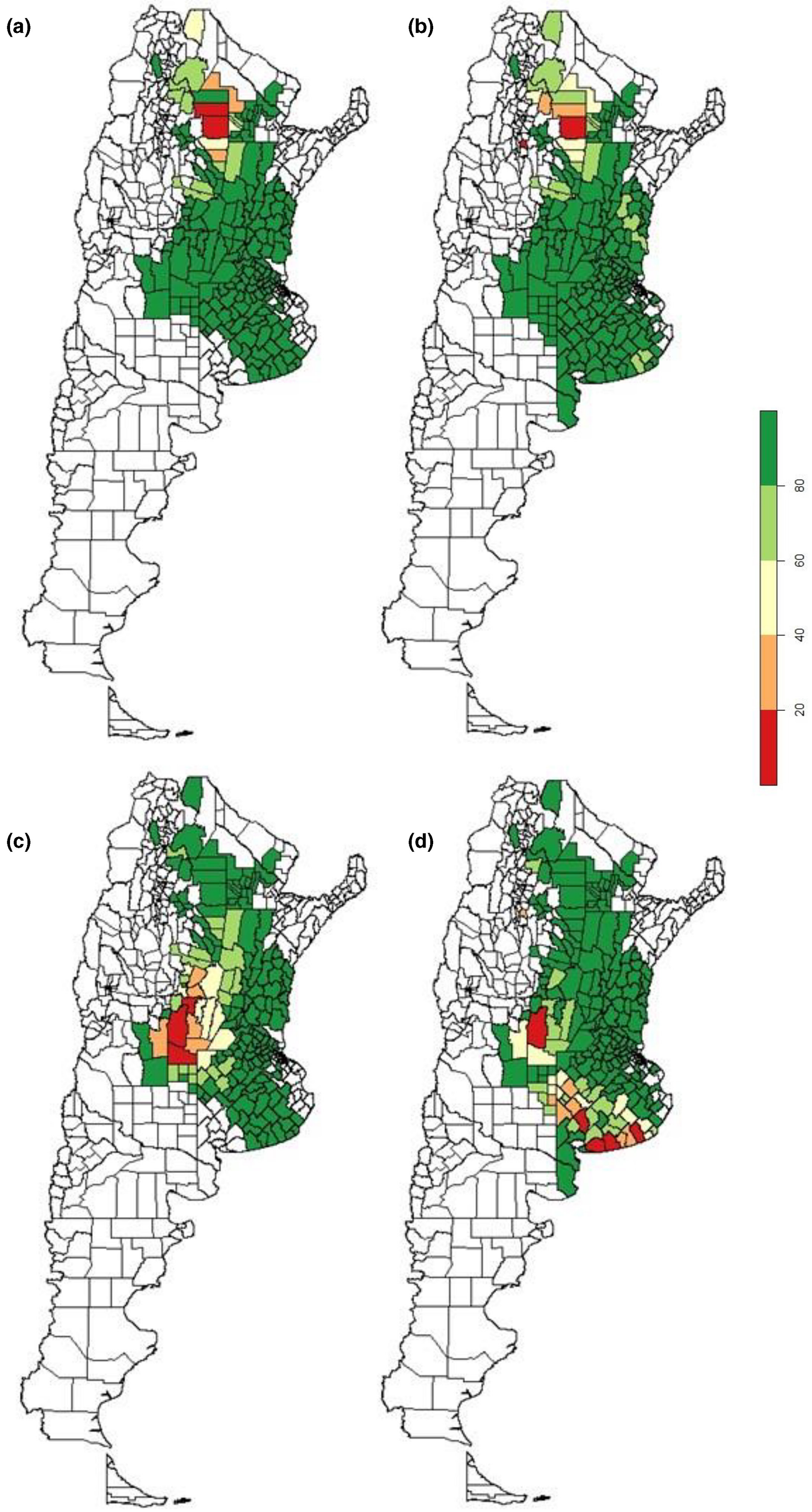

FIGURE 5 Cumulative sum of risk scores for forest loss for (a) the EU, (b) China, and endemic species for (c) the EU and (d) China.

4 | DISCUSSION

The biodiversity scores show considerable variation across space in terms of where the highest risks for species could be for each biodiversity metric compared to forest loss (Figure 2). This is then reflected from the perspective of trade-linked risk in the exemplar shown of Argentinian soy trade. The cumulative curves indicate that a relatively small proportion of biodiversity risk would be incorporated into commitments targeting solely the highest areas of forest loss risk, for example only 8% endemic species risk for EU trade of soy produced in Argentina (Figure 4, Table 1). There is considerable variation between forest loss and other biodiversity metrics in terms

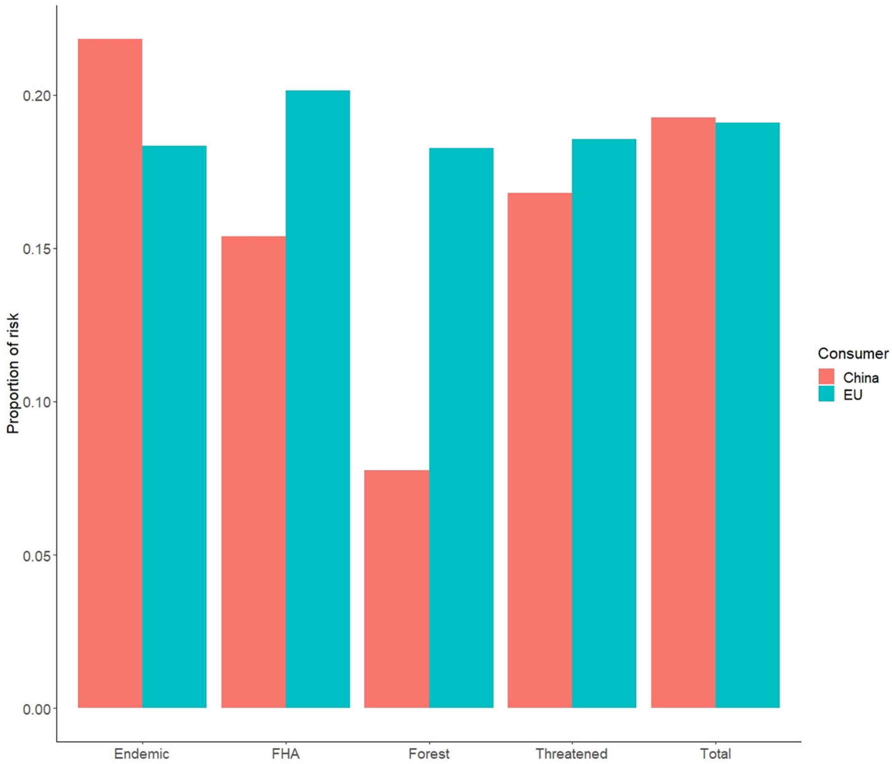

FIGURE 6 Normalised impact scores across five metrics and variation between scores for China and EU trade of soy with Argentina.

of which departments or municipalities are contributing the greatest risk. For example, with EU imports of Argentinian soy, the departments with the highest impacts for forest loss are located in the north of the country whereas for the species-based metrics, there is more congruence and the impacts are concentrated in the centre of Argentina (Figure S1). Biodiversity risks will vary even within forested areas both across landscapes and with the indicator chosen to represent biodiversity (Hill et al., 2019), hence using forest loss alone as a proxy for biodiversity risks overlooking areas of importance. The specific aspect of biodiversity that policy makers or supply chain actors are interested in could lead to grossly different perspectives of risk, even when only considering species related biodiversity risk as demonstrated by the substantial difference in our findings between these metrics and forest loss risk. Consequently, different decisions could be made by supply chain actors in terms of where to focus efforts of reducing risk to biodiversity.

4.1 | Is there congruence of risks across metrics and landscapes?

Sourcing patterns will inevitably lead to different hotspots of risk for different importers. However, there is more congruence with forest loss risk between Chinese and EU trade compared to biodiversity risk metrics. Both importers show the highest impacts in the north of Argentina for forest loss risk, yet for the four biodiversity metrics, there are different patterns between importers (Figure 5; Figures S1 and S2). For example, for China the departments with the highest impacts are located further southeast of the hotspots shown for trade with the EU. Unlike forest loss, there is very little congruence for each individual biodiversity metric across these two importers, with considerably more spatial variation for China in where the highest risks occur and wider distribution of risk, whereas for the EU risk is more concentrated in fewer departments.

Yet, even within importer trade flows, there is variation between biodiversity metrics despite there being more congruence than compared to forest loss risk. Departments contributing to the highest percentages of biodiversity risk for EU trade are located more centrally than forest loss, however there are only a small handful that are highlighted consistently across all four biodiversity metrics (Figure S1). The same is shown for trade with China, and for both importers the difference is particularly apparent for threatened and endemic species (Figures S1 and S2). This is important to note as commodity production is likely to have a higher impact in areas with high levels of endemism or species sensitivity to threats (Martins & Pereira, 2017; Newbold et al., 2020).

Both the divergence of forest loss from biodiversity metrics and the variance across biodiversity metrics is also seen in Brazil (Tables 1 and 2; Figures S3 and S4). Forest loss does however show more alignment with biodiversity metrics than in Argentina, which suggests that the extent to which forest loss can be used as a proxy for other aspects of biodiversity could vary across different landscapes, (Tables 1 and 2). This is demonstrated further in the case of Paraguay (Figures S5 and S6) which, although has less data available, still for example clearly shows the spatial divergence of forest loss from biodiversity metrics for EU trade (Figure S9). However it shows the strongest alignment with biodiversity metrics when considering how many departments it takes to reach a certain risk threshold

(Tables 1 and 2) which is likely due to the coarseness of the data. Hence, the spatial scale of data could influence the extent to which forest loss could represent biodiversity risk in the context of trade linked risk assessments.

4.2 | What are the implications for biodiversity conservation?

The lack of congruence between forest loss and biodiversity metrics, and the inconsistency of alignment between the biodiversity metrics themselves across landscapes and importers demonstrates the importance of incorporating multiple biodiversity indicators in risk assessments. The many dimensions of biodiversity cannot be fully captured in just one indicator (Purvis, 2020) and previous global commitments to address biodiversity loss by using a single target have not been met (Butchart et al., 2010). Due to the multidimensional nature of biodiversity, not only is using multiple indicators important (Marquardt et al., 2019) but also is using indicators which are complementary (Marques et al., 2021). For example, using indicators for both ecosystem multi-functionality and species extinction, as they represent two different goals of mitigating impacts on biodiversity; safeguarding the benefits provided to people by preserving ecosystems and minimising irreversible species loss (Purvis, 2020). Although this study does not use an indicator for ecosystem functionality, it does makes a first step towards this as the additional biodiversity metrics representing species risk can be seen as a complement or an addition to a metric focused on a loss of habitat, in this case, forest. This is evident in our results given that our findings show very different areas of risk exposure when comparing forest loss and species-related biodiversity risk.

The reason for monitoring biodiversity will influence the indicator choice, and different biodiversity indicators are needed to represent various different objectives of biodiversity conservation. For example, preventing species extinctions and preserving the ecosystem services people gain from nature will have separate, distinct requirements (Purvis, 2020). This could therefore mean prioritisation and protection of different areas based on the aspect of biodiversity of interest or the objective of biodiversity conservation. For example, local communities in tropical regions are often highly dependent on biodiversity for the provision of ecosystem services (Dasgupta, 2021), many of which are irreplaceable yet are declining worldwide (IPBES, 2019). However biodiversity targets usually focus more on preventing species extinctions, an indicator that is easily communicable to both policy and public audiences (Butchart et al., 2010; Rounsevell et al., 2020). Given that our results using forest loss and species-related metrics show differences in areas of risk, it is expected that using metrics which represent very different aspects of biodiversity such as ecosystem functionality would show even less alignment in terms of risk exposure, which could present a conflict of interests between different actors.

Preventing forest loss has been a long-term conservation priority as evidence shows deforestation increases the risk of a species becoming threatened (Betts et al., 2017). Biodiversity conservation is often an implicit or explicit goal of attempts to reduce deforestation, however some studies suggest that environmental data is a poor surrogate for biodiversity, especially when compared to using cross-taxon surrogates for conservation planning (Rodrigues & Brook, 2007). Biodiversity values are also inconsistent across different types of forest (Gardner et al., 2009). One such comparison shows that values between forest habitats are highly variable and strongly dependent on the choice of taxa and metric used (Barlow et al., 2007). It demonstrates that there are strong differences in community structure and composition across different forest types and a lack of congruence between taxa in response to land use change patterns. Hence as demonstrated by our results, using forest loss alone as a proxy for other aspects of biodiversity is likely to be inadequate.

Given that conservation science can aid policy decision making by providing information about biodiversity values of different habitats, the variance within one type of habitat is an important observation. The four biodiversity metrics we use in this study are based on the same input data from IUCN and Birdlife, and are therefore are more likely to show congruence than the variety of metrics that are potentially available that use alternative data inputs and methodologies. Thus, given the amount of variance in our results—obtained by just using simple derivations of a species-richness based risk indicator, this suggests a strong likelihood that using more complex metrics will show further divergence. Whilst this study therefore represents a step forward in exploring the implications of the application of alternative biodiversity metrics for deforestation-linked (or indeed other) supply chains, the fact that there appears to be relatively little work being done to explore relationships between more complex indicators within fine scale, trade linked assessments on biodiversity is a cause for conservation concern.

4.3 | What are the implications for policy?

From the perspective of a company or country sourcing materials with links to severe environmental impacts, the ability to conduct a rapid risk assessment is vital to focus efforts on the places of highest concern. In the context of soy trade, there are active efforts to do this for deforestation-risk. For example, the Soft Commodities Forum (a collective of major trading actors) conduct risk assessments for their sourcing in the Cerrado biome (WBCSD, 2022) and the incoming EU due diligence regulation proposes use of a risk-based benchmarking system (EPRS, 2022). In the case of the Soft Commodities Forum activities, the risk assessment has been determined to be ‘the most effective way to drive progress’ and is used to determine where priorities exist for company disclosure, and to support farm-level traceability efforts and investment in local sustainability projects (WBCSD, 2022). In the case of the EU regulation, the risk assessment would determine the reporting requirements for companies subject to this legislation and the level of scrutiny placed on regions of production (European Commission, 2021).

There are broadly two potential responses by actors to the exposure of sourcing associated with high-risk. On the one hand, supply chain actors may aim to mitigate this risk in regions of production by focussing investments in sustainability schemes which aim to reduce their total risk exposure. Risk mitigation could also be achieved by undertaking more granular traceability to determine that their sourcing takes place with low impact even within places of high overall risk. Alternatively, supply chain actors may determine that risk is too high to continue sourcing from these areas, and will seek thereafter alternative sources, which EU proposals indicate is likely the most viable option for small businesses (European Commission, 2021). Encouragement to invest in places of currently unsustainable production could therefore provide opportunities for sourcing patterns to be 'net positive' for forests and associated biodiversity.

However, if supply chain actors decided to shift production, our analysis demonstrates that any assessment of risk based on deforestation is unlikely to be an adequate proxy for broader concerns linked to biodiversity loss. Shifting supply chains will place additional burdens on other regions of production and interventions could induce leakage effects (Meyfroidt et al., 2020; Zu Ermgassen et al., 2020). For example, crop expansion and land use change could instead occur in different regions (Gibbs et al., 2015) or in other habitats which also have a high biodiversity value other than forests, a risk that civil society has been raising as an important gap in the incoming EU legislation (WWF, 2022). Intensification as an attempt to mitigate risks of cropland expansion could also pose greater risks to biodiversity (Phalan et al., 2014). Those prioritising efforts in high-risk landscapes from the perspective of deforestation will only be encompassing in these efforts the portion of the habitats and species currently affiliated with deforestation-risk areas. As shown in this study, this could result in the exclusion of important aspects of biodiversity (Figure 4), as attention will be less on the broader geographies that they are sourcing from. Furthermore, supply chain actors with relatively low existing levels of deforestation risk may still need to account for potential biodiversity risks linked to their current production, which may be overlooked if risk assessments focus solely on deforestation.

If the focus on agri-commodity risk is broadened beyond the scope of deforestation in future, landscape-based approaches and collaboration across multiple countries of consumption will be necessary to ensure that initiatives to promote environmental sustainability have the intended effect. Otherwise, they could simply lead to niche where sustainable products go to one destination with higher sustainability standards, leaving the rest for those with lower sustainability demands (Glasbergen & Schouten, 2015). For any extension to biodiversity, however, the level of alignment on what attributes of biodiversity are most important from the perspective of consuming countries (or supply chain actors) is likely to be fundamental to the success of the alignment of interventions to avoid leakage effects. For example, if one set of actors focuses on protection of threatened species and another on endemic species, then areas determined as being of high risk exposure from the one importer perspective will vary to another.

Furthermore, even if there is alignment on which aspects of biodiversity are important, convergence in focal areas is not guaranteed. For example, for endemic species in Argentina, the focal regions of high risk from the perspective of Chinese sourcing are substantially different from that of the EU (Figure 5). The levels of congruence will also differ depending on the national context. Whilst divergence in the risk profile of consuming countries is inevitable given different sourcing patterns, developing a shared understanding of which aspects of biodiversity are important or should be considered in supply chain risk assessments will be fundamental to understanding potential tensions between the priorities of distinct supply chain actors and/or opportunities for collaboration to protect important landscapes. In the context of efforts to align future multi-lateral policymaking around the areas of highest biodiversity risk, the complexity illustrated in the patterns of risk exposure in our study (Figures 2 and 5) illustrate that further work is necessary to both understand risk exposure in more detail and to formulate adequate responses to the nuanced picture that emerges.

4.4 | What are the limitations and needs for further work?

The application of simple metrics in this analysis means that a number of developments are possible towards an ambition of an increased understanding of trade-linked biodiversity risks. For example, in the case of the forest habitat affiliated species metric utilised, there is an observed lack of congruence with the metric of forest loss. This is likely caused by the use of the IUCN's habitat affiliation information as a proxy for forest dependency. The lack of congruence is therefore not surprising given species often have several habitat affiliations listed by IUCN. Two potential opportunities to build on this work are therefore to develop habitat suitability models (Hill et al., 2019) or use area of habitat to develop biodiversity metrics (Brooks et al., 2019), which could potentially improve congruence between metrics of forest loss and species linked to forests whilst also reducing sampling bias in IUCN/Birdlife data. Listed threats are also not considered in this work, hence taking this into account would also refine metrics. Future work that extends to a broader range of more complex, yet complementary biodiversity indicators is highly recommended to explore the interactions and relationships between different indicators further. However, there are difficulties around choice of indicator given that they are often derived differently, and generation and interpretation of results could become increasingly challenging. For example, IUCN and Birdlife data are updated periodically and the frequency differs across taxonomic groups, whereas indicators based on satellite data generate detailed, immediate information.

The underlying data used for forest loss (Hansen et al., 2013), although the only global dataset available at fine scale resolution, also has its limitations. For example, it defines forest as 'all vegetation taller than 5 m in height', which makes forests indistinguishable from plantations (Tropek et al., 2014). It is therefore important to note that there will be forested areas and forest dependent species in areas such as the Cerrado that are not captured as forest by the Hansen dataset due to the canopy not being high enough or pixels not consisting of the minimum coverage. These areas are likely classified as forests in national maps, hence using a wider definition of deforestation and comparing to the same biodiversity metrics is suggested for further work. Using a national map, such as Brazil's PRODES product for example, could provide further insights in a national context. Although additional forested areas could be incorporated, past studies suggest its estimates of deforestation rates are lower than for the Hansen dataset (Milodowski et al., 2017). For this study though, we justify our use of Hansen data due to its consistency across landscapes and the data being widely adopted by the research community.

Given that the data layers we have used in this analysis are global in scale, methods would be applicable to other regions and crops of interest. It will therefore also be worthwhile extending these analyses to both other crop production systems as well as different landscapes. We also do not consider the intactness of forests, and forest degradation, as opposed to forest loss, can also be associated with high rates of biodiversity loss (Barlow et al., 2016). Furthermore, intensity of crop production is also not considered and both of these limitations are worthy of further exploration. Whilst soy is typically grown in highly-intensive production systems across Latin America (Song et al., 2021), differences in agricultural inputs and production practices will also have important interactions with risk to biodiversity (Dullinger et al., 2021), which warrant further exploration in indicator inter-comparisons linked to trade.

5 | CONCLUSIONS

In conclusion, our analysis demonstrates the need for further work in understanding the relationships between different aspects of biodiversity and their consequences for trade-linked supply chain risk assessment. Given that the metrics we use show a high level of variance despite being subsets of the same data, it is expected that more complex ways of measuring biodiversity will show further divergence, which has implications both for trade policy and biodiversity conservation. Basing conservation strategies or policies on forest loss alone, which has historically been the focus, will likely exclude important aspects of biodiversity, with important potential consequences for supply chain management and sustainable development efforts. Therefore, efforts to 'clean' supply chains and enhance their sustainability may be unsuccessful if other indicators are not considered. Yet, it is important to explore further the relationships between different, more complex biodiversity indicators and forest loss in order to fully understand their interactions and the resultant conclusions for supply chain decision making.

AUTHOR CONTRIBUTIONS

Amy Molotoks, Jonathan Green and Chris West contributed to the study concept and design. Amy Molotoks led the analysis with technical input from Yunxia Wang and Vivian Ribeiro. Amy Molotoks also led writing the article with contributions from Jonathan Green, Chris West and Vivian Ribeiro. All authors have contributed to revising the article and approving the final manuscript.

ACKNOWLEDGEMENTS

Simon Croft is acknowledged for his input for understanding how the Trase model data is produced and how it should be applied. Osvaldo Pereira is also acknowledged for his advice on creating and presenting the cumulative risk scores developed in this paper.

FUNDING INFORMATION

This work is funded by the Global Challenges Research Fund Trade, Development and the Environment Hub project (ES/S008160/1). It is also supported by the Gordon and Betty Moore Foundation (7703.01), Norway's International Climate and Forest Initiative (QZA-21/0156) and the Quadrature Climate Foundation (01-21-000098).

CONFLICT OF INTEREST STATEMENT

The authors have no conflicts of interest to declare.

DATA AVAILABILITY STATEMENT

Underlying data are available under the terms of the Creative Commons Attribution 4.0 International licence (CC-BY 4.0) in the following open access data repository: https://doi.org/10.5281/zenodo.7547812 (Molotoks et al., 2023).

ORCID

Amy Molotoks https://orcid.org/0000-0002-1168-1580
Chris West https://orcid.org/0000-0002-5091-6514

REFERENCES

Barlow, J., Gardner, T. A., Araujo, I. S., Ávila-Pires, T. C., Bonaldo, A. B., Costa, J. E., Esposito, M. C., Ferreira, L. V., Hawes, J., Hernandez, M. I. M., Hoogmoed, M. S., Leite, R. N., Lo-Man-Hung, N. F., Malcolm, J. R., Martins, M. B., Mestre, L. A. M., Miranda-Santos, R., Nunes-Gutjahr, A. L., Overal, W. L., … Peres, C. A. (2007). Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. *Proceedings of the National Academy of Sciences of the United States of America*, *104*(47), 18555–18560. https://doi.org/10.1073/PNAS.0703333104

Barlow, J., Lennox, G. D., Ferreira, J., Berenguer, E., Lees, A. C., Nally, R. M., Thomson, J. R., Ferraz, S. F. B., Louzada, J., Oliveira, V. H. F., Parry, L., Ribeiro de Castro Solar, R., Vieira, I. C. G., Aragão, L. E. O. C., Begotti, R. A., Braga, R. F., Cardoso, T. M., de Oliveira, R. C., Jr., Souza, C. M., Jr., … Gardner, T. A. (2016). Anthropogenic disturbance in tropical forests can double biodiversity loss from deforestation. *Nature*, *535*(7610), 144–147. https://doi.org/10.1038/nature18326

Betts, M. G., Wolf, C., Ripple, W. J., Phalan, B., Millers, K. A., Duarte, A., Butchart, S. H. M., & Levi, T. (2017). Global forest loss disproportionately erodes biodiversity in intact landscapes. *Nature*, *547*(7664), 441–444. https://doi.org/10.1038/NATURE23285

Birdlife. (2021). *Bird species distribution maps of the world*. Birdlife International: NatureServe. https://www.birdlife.org/datazone

Brooks, T. M., Pimm, S. L., Akçakaya, H. R., Buchanan, G. M., Butchart, S. H. M., Foden, W., Hilton-Taylor, C., Hoffmann, M., Jenkins, C. N., Joppa, L., Li, B. V., Menon, V., Ocampo-Peñuela, N., & Rondinini, C. (2019). Measuring terrestrial area of habitat (AOH) and its utility for the IUCN red list. *Trends in Ecology & Evolution*, *34*(11), 977–986. https://doi.org/10.1016/J.TREE.2019.06.009

Butchart, S. H. M., Walpole, M., Collen, B., van Strien, A., Scharlemann, J. P. W., Almond, R. E. A., Baillie, J. E. M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K. E., Carr, G. M., Chanson, J., Chenery, A. M., Csirke, J., Davidson, N. C., Dentener, F., Foster, M., Galli, A., … Watson, R. (2010). Global biodiversity: Indicators of recent declines. *Science*, *328*(5982), 1164–1168. https://doi.org/10.1126/SCIENCE.1187512/SUPPL_FILE/BUTCHART_SOM.PDF

Condit, R., Pitman, N., Leigh, E. G., Jr., Chave, J., Terborgh, J., Foster, R. B., Núñez, P., Aguilar, S., Valencia, R., Villa, G., Muller-Landau, H. C., Losos, E., & Hubbell, S. P. (2002). Beta-diversity in tropical Forest trees. *Science*, *295*(5555), 666–669.

Curtis, P. G., Slay, C. M., Harris, N. L., Tyukavina, A., & Hansen, M. C. (2018). Classifying drivers of global forest loss. *Science*, *361*(6407), 1108–1111. https://doi.org/10.1126/science.aau3445

Dasgupta, P. (2021). The economics of biodiversity: The Dasgupta review. *Journal of Political Ecology*, *28*. https://doi.org/10.2458/jpe.2289

Davies, T. J., & Cadotte, M. W. (2011). Quantifying biodiversity: Does it matter what we measure? In *Biodiversity hotspots* (pp. 43–60). https://doi.org/10.1007/978-3-642-20992-5

De Baan, L., Curran, M., Rondinini, C., Visconti, P., Hellweg, S., & Koellner, T. (2015). High-resolution assessment of land use impacts on biodiversity in life cycle assessment using species habitat suitability models. *Environmental Science and Technology*, *49*(4), 2237–2244. https://doi.org/10.1021/es504380t

Defries, R. S., Rudel, T., Uriarte, M., & Hansen, M. (2010). Deforestation driven by urban population growth and agricultural trade in the twenty-first century. *Nature Geoscience*, *3*(3), 178–181. https://doi.org/10.1038/ngeo756

Delzeit, R., Zabel, F., Meyer, C., & Václavík, T. (2017). Addressing future trade-offs between biodiversity and cropland expansion to improve food security. *Regional Environmental Change*, *17*(5), 1429–1441. https://doi.org/10.1007/s10113-016-0927-1

Department for Environment, Food and Rural Affairs (DEFRA). (2021). *Implementing due diligence on forest risk commodities: Consultation document*. https://consult.defra.gov.uk/international-biodiversity-and-climate/implementing-due-diligence-forest-risk-commodities/supporting_documents/implementingduediligenceconsultationdocument.pdf

Dreoni, I., Matthews, Z., & Schaafsma, M. (2022). The impacts of soy production on multi-dimensional well-being and ecosystem services: A systematic review. *Journal of Cleaner Production*, *335*, 130182. https://doi.org/10.1016/J.JCLEPRO.2021.130182

Dullinger, I., Essl, F., Moser, D., Erb, K., Haberl, H., & Dullinger, S. (2021). Biodiversity models need to represent land-use intensity more comprehensively. *Global Ecology and Biogeography*, *30*(5), 924–932. https://doi.org/10.1111/GEB.13289

European Commission. (2019). *Deforestation and forest degradation*. https://ec.europa.eu/environment/forests/deforestation.htm#:~:text=Deforestation%20and%20forest%20degradation%20negatively,and%20the%20rule%20of%20law

European Commission. (2021). *Proposal for a regulation on deforestation-free products*. https://ec.europa.eu/environment/publications/proposal-regulation-deforestation-free-products_en

European Parliamentary Research Service (EPRS). (2022). *Towards deforestation-free commodities and products in the EU*. https://www.europarl.europa.eu/RegData/etudes/BRIE/2022/698925/EPRS_BRI(2022)698925_EN.pdf

FAO. (2022). *FAOSTAT statistical database*. http://faostat.fao.org/

Ferrier, S. (2002). Mapping spatial pattern in biodiversity for regional conservation planning: Where to from here? *Systematic Biology*, *51*(2), 331–363.

Gardner, T. A., Barlow, J., Chazdon, R., Ewers, R. M., Harvey, C. A., Peres, C. A., & Sodhi, N. S. (2009). Prospects for tropical forest biodiversity in a human-modified world. *Ecology Letters*, *12*(6), 561–582. https://doi.org/10.1111/J.1461-0248.2009.01294.X

Garrett, R., & Rueda, X. (2018). *Measuring impacts of supply chain initiatives for conservation: Focus on Forest-risk food commodities*. *Meridian Institute*.

Garrett, R. D., Levy, S., Carlson, K. M., Gardner, T. A., Godar, J., Clapp, J., Dauvergne, P., Heilmayr, R., le Polain de Waroux, Y., Ayre, B., Barr, R., Døvre, B., Gibbs, H. K., Hall, S., Lake, S., Milder, J. C., Rausch, L. L., Rivero, R., Rueda, X., … Villoria, N. (2019). Criteria for effective zero-deforestation commitments. *Global Environmental Change*, *54*, 135–147. https://doi.org/10.1016/j.gloenvcha.2018.11.003

Gibbs, H. K., Rausch, L., Munger, J., Schelly, I., Morton, D. C., Noojipady, P., Soares-Filho, B., Barreto, P., Micol, L., & Walker, N. F. (2015). Brazil's soy moratorium: Supply-chain governance is needed to avoid deforestation. *Science*, *347*(6220), 377–378. https://doi.org/10.1126/SCIENCE.AAA0181

Gibbs, H. K., Ruesch, A. S., Achard, F., Clayton, M. K., Holmgren, P., Ramankutty, N., & Foley, J. A. (2010). Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. *Proceedings of the National Academy of Sciences of the United States of America*, *107*(38), 16732–16737. https://doi.org/10.1073/pnas.0910275107

Glasbergen, P., & Schouten, G. (2015). Transformative capacities of global private sustainability standards: A reflection on scenarios in the field of agricultural commodities. *Journal of Corporate Citizenship*, *58*, 85–101. https://doi.org/10.2307/jcorpciti.58.85

Godar, J., Persson, U. M., Tizado, E. J., & Meyfroidt, P. (2015). Towards more accurate and policy relevant footprint analyses: Tracing fine-scale socio-environmental impacts of production to consumption. *Ecological Economics*, *112*, 25–35. https://doi.org/10.1016/J.ECOLECON.2015.02.003

Government Publishing Office (GPO). (2021). *Fostering Overseas Rule of law and Environmentally Sound Trade Act of 2021*. https://www.govinfo.gov/app/details/BILLS-117s2950is/related

Green, J., Croft, S. A., Durán, A. P., Balmford, A. P., Burgess, N. D., Fick, S., Gardner, T. A., Godar, J., Suavet, C., Virah-Sawmy, M., Young, L. E., & West, C. D. (2019). Linking global drivers of agricultural trade to on-the-ground impacts on biodiversity. *Proceedings of the National Academy of Sciences of the United States of America*, *116*(46), 23202–23208. https://doi.org/10.1073/pnas.1905618116

Grenyer, R., Orme, C. D. L., Jackson, S. F., Thomas, G. H., Davies, R. G., Davies, T. J., Jones, K. E., Olson, V. A., Ridgely, R. S., Rasmussen, P. C., Ding, T. S., Bennett, P. M., Blackburn, T. M., Gaston, K. J., Gittleman, J. L., & Owens, I. P. F. (2006). LETTERS global distribution and conservation of rare and threatened vertebrates. *Nature*, *444*(7115), 93–96. https://doi.org/10.1038/nature05237

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., Thau, D., Stehman, S. V., Goetz, S. J., Loveland, T. R., Kommareddy, A., Egorov, A., Chini, L., Justice, C. O., & Townshend, J. R. G. (2013). High-resolution global maps of 21st-century Forest cover change. *Science*, *134*(November), 850–854.

Henders, S., Persson, U. M., & Kastner, T. (2015). Trading forests: Land-use change and carbon emissions embodied in production and exports of forest-risk commodities. *Environmental Research Letters*, *10*(12). https://doi.org/10.1088/1748-9326/10/12/125012

Hill, S. L. L., Arnell, A., Maney, C., SHM, B., Hilton-Taylor, C., Ciciarelli, C., Davis, C., Dinerstein, E., Purvis, A., & Burgess, N. D. (2019). Measuring Forest biodiversity status and changes globally. *Frontiers in Forests and Global Change*, *1*, 70. https://doi.org/10.3389/ffgc.2019.00070

IPBES. (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*. I. S. Díaz, J. Settele, E. S. Brondízio, H. T. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, K. A. Brauman, S. H. M. Butchart, K. M. A. Chan, L. A. Garibaldi, K. Ichii, J. Liu, S. M. Subramanian, G. F. Midgley, P. Miloslavich, Z. Molnár, & D. Obura (Eds.). IPBES. https://zenodo.org/record/3553579#.YfmYTerMI2w

IUCN. (2021). *The IUCN Red List of Threatened Species*. Version 2021-3. https://www.iucnredlist.org

Joppa, L. N., O'Connor, B., Visconti, P., Smith, C., Geldmann, J., Hoffmann, M., Watson, J. E. M., Butchart, S. H. M., Virah-Sawmy, M., Halpern, B. S., Ahmed, S. E., Balmford, A., Sutherland, W. J., Harfoot, M., Hilton-Taylor, C., Foden, W., Minin, E. D., Pagad, S., Genovesi, P., … Burgess, N. D. (2016). Filling in biodiversity threat gaps: Only 5% of global threat data sets meet a 'gold standard'. *Science*, *352*(6284), 416–418. https://doi.org/10.1126/SCIENCE.AAF3565

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

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? *Ecological Indicators*, *103*(April), 461–470. https://doi.org/10.1016/j.ecolind.2019.04.022

Marques, A., Robuchon, M., Hellweg, S., Newbold, T., Beher, J., Bekker, S., Essl, F., Ehrlich, D., Hill, S., Jung, M., Marquardt, S., Rosa, F., Rugani, B., Suárez-Castro, A. F., Silva, A. P., Williams, D. R., Dubois, G., & Sala, S. (2021). A research perspective towards a more complete biodiversity footprint: A report from the world biodiversity forum. *The International Journal of Life Cycle Assessment*, *26*, 238–243. https://doi.org/10.1007/s11367-020-01846-1

Marshall, E., Wintle, B. A., Southwell, D., & Kujala, H. (2019). What are we measuring? A review of metrics used to describe biodiversity in offsets exchanges. *Biological Conservation*, *241*, 108250. https://doi.org/10.1016/j.biocon.2019.108250

Martins, I., & Pereira, H. M. (2017). Improving extinction projections across scales and habitats using the countryside species-area relationship. *Scientific Reports*, *7*, 12899. https://doi.org/10.1038/s41598-017-13059-y

Meyfroidt, P., Börner, J., Garrett, R., Gardner, T., Godar, J., Kis-Katos, K., Soares-Filho, B. S., & Wunder, S. (2020). Focus on leakage and spillovers: Informing land-use governance in a tele-coupled world. *Environmental Research Letters*, *15*(9), 090202. https://doi.org/10.1088/1748-9326/ab7397

Milodowski, D. T., Mitchard, E. T. A., & Williams, M. (2017). Forest loss maps from regional satellite monitoring systematically underestimate deforestation in two rapidly changing parts of the Amazon. *Environmental Research Letters*, *12*(9), 094003. https://doi.org/10.1088/1748-9326/aa7e1e

Molotoks, A., Green, J., Ribeiro, V., Wang, Y., & West, C. (2023). Assessing the value of biodiversity-specific footprinting metrics linked to South American soy trade. *Zenodo*, https://doi.org/10.5281/zenodo.7547812

Naidoo, R., & Iwamura, T. (2007). Global-scale mapping of economic benefits from agricultural lands: Implications for conservation priorities. *Biological Conservation*, *140*(1–2), 40–49. https://doi.org/10.1016/j.biocon.2007.07.025

Newbold, T., Oppenheimer, P., Etard, A., & Williams, J. J. (2020). Tropical and Mediterranean biodiversity is disproportionately sensitive to land-use and climate change. *Nature Ecology & Evolution*, *4*, 1630–1638. https://doi.org/10.1038/s41559-020-01303-0

Pendrill, F., Persson, U. M., Godar, J., & Kastner, T. (2019). Deforestation displaced: Trade in forest-risk commodities and the prospects for a global forest transition. *Environmental Research Letters*, *14*(5). https://doi.org/10.1088/1748-9326/ab0d41

Phalan, B., Green, R., & Balmford, A. (2014). Closing yield gaps: Perils and possibilities for biodiversity conservation. *Philosophical Transactions of the Royal Society, B: Biological Sciences*, *369*, 20120285. https://doi.org/10.1098/rstb.2012.0285

Purvis, A. (2020). A single apex target for biodiversity would be bad news for both nature and people. *Nature Ecology & Evolution*, *4*(6), 768–769. https://doi.org/10.1038/s41559-020-1181-y

Rodrigues, A. S. L., & Brook, T. M. (2007). Shortcuts for biodiversity conservation planning: The effectiveness of surrogates. *Ecology, Evolution, and Systematics*, *38*, 713–737. https://doi.org/10.1146/annurev.ecolsys.38.091206.095737

Rounsevell, M. D. A., Harfoot, M., Harrison, P. A., Newbold, T., Gregory, R. D., & Mace, G. M. (2020). A biodiversity target based on species extinctions. *Science*, *368*(6496), 1193–1195. https://doi.org/10.1126/science.aba6592

Smit, L., Bright, C., Mccorquodale, R., Bauer, M., Deringer, H., Baeza Breinbauer, D., Torres-Cortés, F., Alleweldt, F., Kara, S., Salinier, C., & Tobed, H. (2020). *Study on due diligence requirements through the supply chain*. Publications Office of the European Union.

Song, X. P., Hansen, M. C., Potapov, P., Adusei, B., Pickering, J., Adami, M., Lima, A., Zalles, V., Stehman, S. V., di Bella, C. M., Conde, M. C., Copati, E. J., Fernandes, L. B., Hernandez-Serna, A., Jantz, S. M., Pickens, A. H., Turubanova, S., & Tyukavina, A. (2021). Massive soybean expansion in South America since 2000 and implications for conservation. *Nature Sustainability*, *4*(9), 784–792. https://doi.org/10.1038/s41893-021-00729-z

Strassburg, B. B. N., Brooks, T., Feltran-Barbieri, R., Iribarrem, A., Crouzeilles, R., Loyola, R., Latawiec, A. E., Oliveira Filho, F. J. B., de Scaramuzza, C. A., Scarano, F. R., Soares-Filho, B., & Balmford, A. (2017). Moment of truth for the Cerrado hotspot. *Nature Ecology & Evolution*, *1*, 99. https://doi.org/10.1038/s41559-017-0099

Tilman, D., Clark, M., Williams, D. R., Kimmel, K., Polasky, S., & Packer, C. (2017). Future threats to biodiversity and pathways to their prevention. *Nature*, *546*(7656), 73–81. https://doi.org/10.1038/nature22900

Transparency for Sustainable Economies (Trase). (2022). *Intelligence for sustainable trade*. https://www.trase.earth/

Tropek, R., Sedláček, O., Beck, J., Keil, P., Musilová, Z., Šímová, I., & Storch, D. (2014). Comment on 'high-resolution global maps of 21st-century forest cover change'. *Science*, *344*(6187). https://doi.org/10.1126/SCIENCE.1248753/ASSET/FC4B61B0-2355-4DF5-872E-D9D1815D107E/ASSETS/GRAPHIC/344_981D_F1.JPEG

World Bank. (2022). *World Development Indicators*. https://data.worldbank.org/

World Business Council for Sustainable Development (WBCSD). (2022). *Soft Commodities Forum Progress Report*. https://wbcsdpublications.org/scf/

World Wide Fund for Nature (WWF). (2022). *Beyond forests: Reducing the EU's footprint on all natural ecosystems*. https://wwfeu.awsassets.panda.org/downloads/beyond_forests_en.pdf

WWF. (2014). *The growth of soy: Impacts and solutions*. WWF International.

Zu Ermgassen, E. K. H. J., Ayre, B., Godar, J., Bastos Lima, M. G., Bauch, S., Garrett, R., Green, J., Lathuillière, M. J., Löfgren, P., MacFarquhar, C., Meyfroidt, P., Suavet, C., West, C., & Gardner, T. (2020). Using supply chain data to monitor zero deforestation commitments: An assessment of progress in the Brazilian soy sector. *Environmental Research Letters*, *15*(3). https://doi.org/10.1088/1748-9326/AB6497

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Figure S1: Cumulative proportion of risk scores within each department in Argentina for trade with the EU.
Figure S2: Cumulative proportion of risk scores within each department in Argentina for trade with China.
Figure S3: Cumulative ranked risk scores by forest loss for EU trade with Brazil.
Figure S4: Cumulative ranked risk scores by forest loss for Chinese trade with Brazil.
Figure S5: Cumulative ranked risk scores by forest loss for EU trade with Paraguay.
Figure S6: Cumulative ranked risk scores by forest loss for Argentinian trade with Paraguay.
Figure S7: Cumulative proportion of risk scores within each municipality in Brazil for trade with the EU.
Figure S8: Cumulative proportion of risk scores within each municipality in Brazil for trade with China.
Figure S9: Cumulative proportion of risk scores within each department in Paraguay for trade with the EU.
Figure S10: Cumulative proportion of risk scores within each department in Paraguay for trade with Argentina.
Figure S11: Normalised risk scores across five metrics and variation between scores for Chinese and EU trade of soy with Brazil.
Figure S12: Normalised risk scores across five metrics and variation between scores for Argentinian and EU trade of soy with Paraguay.

How to cite this article: Molotoks, A., Green, J., Ribeiro, V., Wang, Y., & West, C. (2023). Assessing the value of biodiversity-specific footprinting metrics linked to South American soy trade. *People and Nature*, *00*, 1–16. https://doi.org/10.1002/pan3.10457