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Vliv české spotřeby na globální stopu biodiverzity

The biodiversity footprint of consumption in the Czech Republic

Diplomová práce

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Prohlášení

Prohlašuji, že jsem závěrečnou práci zpracoval samostatně a že jsem uvedl všechny použité informační zdroje a literaturu. Tato práce ani její podstatná část nebyla předložena k získání jiného nebo stejného akademického titulu.

V Praze dne ....

Podpis

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Abstract

The exponential increase in the scale of human influence on Earth, especially in the past century, has now led to a crisis of biodiversity. Most direct drivers of the biodiversity crisis are present at local scales – local ecosystems are destroyed by a land use change, local populations are annihilated by overhunting. Nevertheless, in the globalized world, our actions do not only affect the ecosystems directly around us but also those located on the other side of the globe. Next to the direct impacts we pose on the ecosystems we live in, we are also partly responsible for the indirect, tele-coupled impacts we pose on distant ecosystems. The goal of this study is to map the impacts consumers in the Czech Republic cause to ecosystems in other countries. Environmental footprints, and specifically biodiversity footprint, are the tools commonly used to measure such distant environmental impacts and to allocate responsibility for them to the final consumers. There are currently multiple methodologies to quantify biodiversity footprint which are, nevertheless, burdened by some methodological problems or do not properly fit the goal of this study. Therefore, a novel method was developed here employing the Biodiversity Intactness Index as the measure of the state of ecosystems. This biodiversity footprint, or more precisely "biodiversity-intactness extended land footprint," indicates the state of an ecosystem as a result of production of goods and allocates the responsibility for this state to the final consumers. Data on international trade of the Czech Republic between the yeas 1995-2015 were analyzed using this novel method. The largest biodiversity footprints were coupled with products imported from the neighboring European countries, but also from tropical countries like Côte d'Ivoire or Indonesia. The largest imported biodiversity footprint flows were coupled with products of forestry, vegetables, fruits, nuts, and other crops. The flux of imported biodiversity footprint increased nearly six-fold over the assessed period. Nevertheless, the biodiversity footprint of products exported from the Czech Republic grew even faster, the Czech Republic became a net exporter of biodiversity footprint by the end of the period. A pattern of exploitation of "developing" countries by the "developed" is apparent for the Czech Republic as well, but it is significantly weaker than what was identified in other studies. A major inter-annual variability in the countries of origin of the imported goods and in the sizes of the biodiversity footprint coupled with each product category indicates that conclusions derived only from single-year data – which was so far a common practice - might be misleading.

Keywords

Biodiversity footprint, international trade, tele-coupling, Czech consumption

Abstrakt

Exponenciální nárůst lidského vlivu na naši planetu, zejména v minulém století, vedl až k současné krizi biologické rozmanitosti. Většina přímých příčin krize biodiversity se projevuje lokálně– místní ekosystémy jsou ničeny změnou ve využívání krajiny, lokální populace jsou hubeny nadměrným lovem. Nicméně v současném globalizovaném světě neovlivňujeme pouze přímo ekosystémy kolem nás, ale také ekosystémy umístěny na druhé straně zeměkoule. Vedle přímých dopadů, které způsobujeme ekosystémům, ve kterých žijeme, jsme tedy také částečně zodpovědní za nepřímé, tele-propojené dopady na vzdálené ekosystémy. Cílem této studie je zmapovat dopady spotřebitelů v České republice na ekosystémy v jiných zemích. Environmentální stopy, a zejména stopa biodiversity, jsou nástroje běžně používané k měření těchto vzdálených dopadů na životní prostředí a k přenesení odpovědnosti za ně na konečné spotřebitele. V současné době existuje několik metodik pro kvantifikaci stopy biodiversity, které jsou nicméně zatíženy některými metodologickými problémy nebo neodpovídají cíli této studie. Proto zde byla vyvinuta nová metoda využívající index neporušenosti biologické rozmanitosti (Biodiversity Intactness Index) jako měřítko stavu ekosystémů. Tato stopa biodiversity, nebo přesněji "stopa na krajinu vážená neporušeností biodiversity", vyjadřuje stav ekosystému v důsledku výroby zboží a přiděluje odpovědnost za tento stav konečným spotřebitelům. Pomocí této metodiky byla analyzována data o mezinárodním obchodu České republiky v letech 1995-2015. Největší stopy biodiversity byly spojeny s produkty dováženými ze sousedních evropských zemí, ale také z tropických zemí, jako je Pobřeží slonoviny nebo Indonésie. Největší dovážené toky stopy biodiversity byly spojeny s produkty lesnictví, zeleninou, ovocem, ořechy a s dalšími plodinami. Velikost toku importované stopy biodiversity se během posuzovaného období zvýšila téměř šestinásobně, stopa produktů vyvážených z České republiky však rostla ještě rychleji a Česká republika se ke konce analyzovaného období stala čistým vývozcem stopy biodiversity. Vzorec vykořisťování "rozvojových" zemí "rozvinutými" zeměmi je patrný také pro Českou republiku, ale je výrazně slabší, než jak byl odhalen v jiných studiích. Velká meziroční variabilita zemí původu dováženého zboží a velikosti stopy jednotlivých kategorií výrobků naznačuje, že závěry odvozené pouze z jednoletých údajů – což byla dosud běžná praxe – by mohly být zavádějící.

Klíčová slova

Stopa biodiversity, mezinárodní obchod, vzdálená propojení, Česká spotřeba

Contents

1 Introduction

Throughout the history of Earth evolved millions of species of animals, plants, fungi, or bacteria who form the diversity of Life. With this variety of life comes also a great variety of survival strategies the species assume. We humans are but one of the millions of species of animals. Yet we are fundamentally different. All animals are directly interconnected with their surrounding ecosystems, where they find all food and shelter. Humans used to, and some still do, live in the same way, sourcing all their food from the surrounding landscape. This has changed radically though, especially over the course of past several hundred years. Only a small fraction of human population now spends their time gathering food, particularly in the "developed" countries. Few people roam forests to gather fruits, nuts, and firewood to bring warmth. Few people struggle on plains to kill other animals for meat. We have tractors, combine harvesters, intense animal farms, and fossil fuels. In cities, we get food from supermarkets, water from pipes, and warmth from central heating systems. It is clear most people lost the direct connection with the surrounding ecosystems, and many people believe that thanks to technology we managed to shed our dependence on nature as well.

It might not be directly apparent when we buy food in a supermarket, but we still need soil, water, nutrients, and sunlight to produce food. We still need grasslands to feed cattle, we still need rain to bring water, we still need trees to produce wood, we still need photosynthesizing organisms to produce oxygen we breathe. Humans are still a part of Earths ecosystems, and still are fully dependent on the gifts they provide. But it is true that the dynamic has changed. Human population rose from around 1 billion at the beginning of the 19th century (Van Bavel 2013) to almost 8 billion people now (*worldometers.info/world-population*). The growth in the scale of human activity in the past century has been so rapid it was dubbed the Great Acceleration (Steffen, Broadgate et al. 2015). The biomass of humans is now one order of magnitude larger than the combined biomass of all wild mammals (Bar-On, Phillips et al. 2018). Humans now appropriate more than 25% of the net primary production of potential vegetation (Krausmann, Erb et al. 2013). All in all, humans are no longer just a part of ecosystems, but are now a dominant force affecting ecosystems on a planetary scale. The influence of humans on Earth is so great it is now widely accepted we live in a new geological epoch of the Anthropocene (Lewis and Maslin 2015). We now have the capacity to disrupt the balance in the functioning of the Earth system - which is currently evident on the changing climate (IPCC 2021) – and to modify or destroy Earths natural ecosystems. Unfortunately, this is exactly what we choose to do. Nevertheless, it is highly dubitable whether human civilization as we know it can survive with the disrupted Earth systems and without functioning natural ecosystems. On the other hand, we also have the potential capacity to *not* destroy and disrupt, to share the space and resources on Earth with other organisms to mutual benefit.

It seems policymakers and the public in some countries and the EU start to acknowledge the need to limit the damage we cause to nature. With policies like European Green Deal and Farm2Fork strategy (*ec.europa.eu*), and with many protected natural areas being established in the past decades, it seems like the current is starting to turn towards a more harmonious co-habitation with nature on the local level. Such a shift seems to be the most apparent in the rich industrialized countries in Europe, where the domination and transformation of landscape by humans was historically the most widespread. This seems to be in line with the Environmental Kuznets Curve hypothesis (Dinda 2004). However, while we were losing the connection with local ecosystems, the interconnectedness of the planet was increasing. The goods we buy in our local supermarket can come from the other side of the country, the other side of the continent, or the other side of the globe. This makes it even harder to grasp the impact we have on nature (Meyfroidt, Bremond et al. 2022). Even local products can have distant tele-coupled impacts on biodiversity, for example: cows raised nearby can be fed by soybeans grown in China or Brazil, and local chocolate is made from cocoa grown in Africa. While it depends on what we consume, and how and where it was produced, there is a clear relationship between affluence (the rate of consumption) and the impacts on nature (Wiedmann, Lenzen et al. 2020), and there is little evidence of absolute decoupling between growth of affluence and biodiversity impacts (Otero, Farrell et al. 2020, Bjelle, Kuipers et al. 2021). Many environmental improvements on the local level were achieved mainly by outsourcing the impact to other (poorer) countries (Wiedmann and Lenzen 2018). Nevertheless, the final consumers of those products are ultimately responsible for the environmental damage coupled with production, no matter its location.

Even well-meant policies can have overall negative impacts if not based on comprehensive knowledge and sound data. It is necessary to uncover and describe, as far as possible, all underlying drivers for any policy designated to halt environmental destruction to be effective. For a policy aimed to stop deterioration of ecosystems and biodiversity it is important to target not only the direct drivers, like land use change or overexploitation of species, but also the underlying socio-economic factors like population, consumption, trade, or corruption (Driscoll, Bland et al. 2018). The goal of this study is to investigate the role of international trade as one of the socio-economic drivers of ecosystem deterioration. Specifically, I assess the consumption-based biodiversity footprint embedded in the products imported to the Czech Republic. By comparing the imported biodiversity footprint to the biodiversity footprint exported, I aim to describe the role and position of the Czech Republic in the global dynamics of nature-exploitation and biodiversity loss.

Although there were already several studies analyzing global biodiversity footprints of nations or the effect of international trade on biodiversity, there is yet no well-established and widely accepted methodology. In my opinion, the approaches taken in those studies are somewhat defective or do not sufficiently serve the goal I have set here. Therefore, I develop here a novel approach to biodiversity footprint calculation that, hopefully, serves the purpose. Nevertheless, in order for the indicator to properly reflect the problem under investigation, I start with a discussion of the Theoretical and contextual background (Section 2), where I also try to summarize the position from which I conduct this study. Biodiversity, and especially biodiversity protection, is a multi-dimensional and value-laden concept. In Section 2.1 I analyze and discuss the various perspectives on what is biodiversity and why (whether) it should be valued. In Section 2.2 I describe the trends of global biodiversity and the most important drivers of biodiversity loss; and in Section 2.3 I discuss what are footprint indicators and analyze the various approaches to biodiversity footprint quantification. Based on those theoretical discussions, I then outline the Materials and methods (Section 3), i.e., the data I used, the way I processed and utilized the data, and the rationale behind those decisions. In Section 4 I describe and analyze the derived results. There, I present the results for the year 2015 (Section 4.1), the evolution of imports of biodiversity footprint between the years 1995 and 2015 (Section 4.2), I analyze in which countries is this footprint located (Section 4.3) and how is the total footprint distributed among product categories (Section 4.4); and, finally, I analyze the trade balance for the Czech Republic as the difference between imports and exports of biodiversity footprint (Section 4.5). In Section 5 I discuss the method and results. I start with Limitations and uncertainty (Section 5.1), then I compare the results of biodiversity footprint with land footprint (Section 5.2), evaluate and discuss the behavior of the newly developed biodiversity footprint indicator (Section 5.3), and outline the direction for future research (Section 5.3). Finally, I present and discuss the conclusions of this study (Section 6).

2 Theoretical and contextual background

2.1 Biodiversity loss

Nature is changing, there is no doubt about it (Sage 2020). While many scientists raise alarm (Ceballos, Ehrlich et al. 2015, Díaz, Settele et al. 2019, Leclère, Obersteiner et al. 2020, Pyšek, Hulme et al. 2020), and for already many decades there have been valiant (albeit too often insufficient) efforts to protect nature and biodiversity, rarely do we pose the question: What does a change of nature mean? or Why do we call it a *loss* of biodiversity? For most people this framing comes naturally. If a species goes extinct, we see it as an irreversible loss of something valuable. If a tropical forest, rich grassland, or coral reefs disappear, we usually see it as a loss of something interesting and beautiful. There are also many less "romantic" reasons to perceive such a disappearance as a loss. But is it really? Periodically there occur voices that challenge this view (e.g., Thomas (2017)). They tend to say (in short): look at the geological history of the Earth, look how many mass extinctions there have been, and life always returns to a rich state; after a massive change and loss, more new species evolve; or, more specifically, look only twenty thousand years back, at the end of the last ice age the Earth looked completely different and most contemporary ecosystems did not even exist; change is a part of life, and "*there is no logic in defining [the] past change as good and natural and at the same time describing more recent and future change as regrettable and unnatural*" (Thomas 2017, p. 96). From a perspective of an uninvolved, uninterested observer this is true. But, even as natural scientists, we are not aliens observing what is happening on Earth from a warm place far away. We live here. Hence, these voices are completely wrong for at least two reasons: 1) the past changes were external and "natural," but today's change is our collective doing and collective decision as humanity, ergo we can decide not to cause the change, which makes every difference; 2) we are humans, so we necessarily take the human perspective (we cannot otherwise). The great changes did happen during the history of the world, *but* it is only in the last epoch (i.e., thanks to the last major changes) that humanity could evolve into what it is now, never before. Thus, from the perspective of humankind (or at least the people who live now), the past changes necessarily are *good* because they enabled us to live. It is only thanks to these changes that natural science can gain knowledge about biodiversity, and only thanks to these changes I can write this text. All we know and love exists today only thanks to these changes (of course, all that we hate too). Conversely, the current and future changes threaten this familiar favorable state of the world, so there is every logic in seeing them as regrettable. Although people have been changing the face of the Earth for thousands of years (Ellis, Gauthier et al. 2021), we now live in the epoch of the Anthropocene, where anthropogenic drivers are becoming stronger than the natural. This fundamental shift implies that it is ultimately up to us to decide what kind of world we want to live in, what kind and "amount" of biodiversity we want to have around us. This is a normative question we need to ask, not a physical external fact given for objective observation.

Hence, it is reasonable to see the change in biological diversity not as something that is just happening but as something problematic and worth the attention. Yet there are various ways to frame this issue that would yield contrasting recommendations - from various values and goals of different actors to differences in what does the word "biodiversity" mean and how should biodiversity be measured. "Biodiversity" is one of the keywords of contemporary environmental research and policy, and like "sustainability" it is used in various contexts and can mean different things for different people. A comprehensive definition of biodiversity could say that it is "*the variety of life, encompassing variation at all levels, from the genes within a species to biologically created habitat within ecosystems*" (Duffy, Amaral-Zettler et al. 2013). In this broad approach to biodiversity there were proposed six classes of essential biodiversity variables: genetic composition, species populations, species traits, community composition, ecosystem structure, and ecosystem function (Pereira, Ferrier et al. 2013). In those classes there can be identified many descriptive variables, however for many of those variables there are significant issues regarding their specific method of measurement and even technical feasibility of data gathering (Schmeller, Weatherdon et al. 2018). Therefore, the broad definition of biodiversity is not very practical or operational for larger-scale studies, and often it is narrowed down to the level of species richness as the most convenient measure (Jarzyna and Jetz 2018, Storch, Šímová et al. 2021). Indeed, the diversity of species is what most people imagine under the term. Still, there are multiple ways to describe species richness (*alpha, beta, gamma*) at different spatial (local, regional, global) and temporal scales (McGill, Dornelas et al. 2015). These differences might be the source of sometimes contradictory results as, for example, alpha diversity on local scale generally shows no significant trend (extinctions even out with colonization), it is increasing on the regional scale (colonization stronger than extinction), but decreasing on the global scale (global homogenization) (McGill, Dornelas et al. 2015). Hence, it cannot be simply said that biodiversity is declining, because it depends on which kind of biodiversity we mean, and on which scale we measure it. The most depressing conclusion about the state of biodiversity says that, based on the rate of species extinction, we have already entered the 6th mass extinction event (Ceballos, Ehrlich et al. 2015). Nevertheless, we are actually still far from having described all the present species – while for mammals probably 97% of species have been described, it is likely less than 30% for arthropods (Pimm and Raven 2019) -, and hence such conclusions can be disputed. On the one hand, they might be too pessimistic, on the other hand the assumed rates of extinction could be severely underestimated, since many species might have gone extinct even before scientists described them, which is called the Linnean shortfall (Hortal, De Bello et al. 2015). This indicates one of the main problems: ecological systems are ”*non-stationary, high-dimensional, nonlinear, stochastic, subject to positive and negative feedback loops, show alternative stable states and hysteresis and a host of other processes which make them endlessly fascinating to study and staggeringly hard to understand fully*” (Clements and Ozgul 2018). The data and knowledge we possess about biological diversity are still very limited in quantity and quality (Jetz, McGeoch et al. 2019), so any strong or catastrophic conclusions are still open to condemnation by skeptics (Leung, Hargreaves et al. 2020). Furthermore, there are several shortfalls and biases inherent in the processes of generating knowledge about nature (Hortal, De Bello et al. 2015) that might strongly affect our conclusions about the state and change of biodiversity.

As described above, the conclusions and policy-recommendations of biodiversity studies are affected by the chosen perspective which is determined by many value-based decisions. For example, the biodiversity trends described by McGill, Dornelas et al. (2015) would be different if non-native “alien” species were not included. The strongly emotional term “alien” that is commonly used for non-native species (Early, Bradley et al. 2016, Pyšek, Hulme et al. 2020) indicates that valuations are deeply embedded even in the allegedly objective natural science. Already the decision of scientists to do research in this field generally implies that they value biodiversity, each in their own way. We could agree with Crist, Mora et al. (2017) that “*whether people value the natural world for its intrinsic standing or for the ecological services it provides humanity [...], sustaining Earth’s biological wealth is an ecumenical good,”* but the way the issue is framed in research and its conclusions and real-world acts would still be determined by the specific valuation mechanisms (Minteer 2009). This is no place for an exhaustive analysis of all views on environmental ethics, but, in my opinion, it would be useful to present several (highly) simplified examples. The ethical views on nature could be broadly divided to anthropocentric, where the only source of value is a human because only humans have moral agency; and non-anthropocentric views, where other parts of nature have an intrinsic value, even without a relation to a human subject (Des Jardins 2012). But which parts of nature have this intrinsic value? In relation to biodiversity protection this could have very different implications. We could see the genetic and species variability as intrinsically valuable (there could be multiple reasons why), which would imply that any loss of this variability is bad, so we need to ensure the survival of a maximum number of species and genes. In a strict sense, though, it would not really matter whether those species survive in their original habitat or in captivity. Contrarily, we could see the will to live, manifested by every organism, as the basis of ethical value. In that case it is morally wrong to just take the life away (whether it could be justified by a greater good is a different question). In a strict sense, it would not really matter if the species would go extinct if all the individuals were allowed to live out their life fully and "happily." In yet another view, not individuals or individual species, but entire ecosystems are the locus of intrinsic value. In this case we would be required to wipe out individuals or even entire (mostly non-native) species if it were necessary to ensure conservation and functioning of an ecosystem, as advocated, for example, by Aldo Leopold (1949 [1989]). There are many more non-anthropocentric approaches to this question, and their combinations. On the other hand there are also multiple ways to value nature from the anthropocentric perspective, probably best exemplified in the concept of ecosystem services (Costanza, d'Arge et al. 1997) or nature's contribution to people (Pascual, Balvanera et al. 2017). Whether productive, regulating, supportive, or cultural services (Millennium Ecosystem Assessment 2005) are preferred would strongly affect the approach to biodiversity protection. Furthermore, the outcome of an anthropocentric valuation (also non-anthropocentric) would be affected by the decision to what degree and how would the interest of future generations be included, notwithstanding there are many other western, non-western, and indigenous cosmologies based on different human-nature relationship models (Muradian and Pascual 2018). What would this mean for studies on the change of biodiversity? First of all, the basic values should be acknowledged, and their effect be reflected. Different scientific-field communities could, legitimately, have different sets of shared values, and there are basically two ways to resolve this: either all get to agree on one set of shared values and one perspective, which is unlikely; or the illusion of a single universal scientific truth is abandoned, and a pluralism of views is embraced (Pascual, Adams et al. 2021). In this work, I will try to follow the second approach of multiple legitimate views on one phenomenon, in the sense I do not see the perspective taken here as the single correct one, just one of many.

2.2 Biodiversity trends and drivers

Since there are multiple perspectives that can be taken when looking at the state and trends of biodiversity, information and conclusions do often vary. Based on the data from the Living

Planet Database (*livingplanetindex.org*) there have been indicated significant declines of vertebrate populations (McRae, Deinet et al. 2017, WWF 2020). Yet other authors with the same data concluded that only a fraction of vertebrate populations is declining, a fraction is increasing, and for most species there is no significant trend as their populations are stable or oscillate (Daskalova, Myers-Smith et al. 2020, Leung, Hargreaves et al. 2020). This would indicate that there is no global catastrophic loss of species, most populations are stable while the winners and losers are in balance (Dornelas, Gotelli et al. 2019). Such conclusions would be rather optimistic because they indicate that we should predominantly focus on the specific declining populations and the causes of their decline, not necessarily look for a systemic change to solve a massive global biodiversity loss. On the other hand, there might be limitations to such conclusions. First, there have been doubts about the methodology used to reach them and about the effect of inherent biases in the data (Loreau, Cardinale et al. 2022, Murali, de Oliveira Caetano et al. 2022). Second, the fact that most populations are currently stable or increasing does not necessarily mean there is no global biodiversity crisis, because for many species the most catastrophic population declines and habitat destruction happened already long before now; a species can have a stable or increasing population trend but still be critically endangered (Mehrabi and Naidoo 2022). Furthermore, biodiversity change is not necessarily coupled to the change in species richness (Hillebrand, Blasius et al. 2018).

There are similarly mixed messages regarding the trend of insect diversity. Several reports of catastrophic declines of insect abundance (Hallmann, Sorg et al. 2017) and diversity (Sánchez-Bayo and Wyckhuys 2019) have been widely covered by media as indications of an "Ecological Armageddon" or "Insectageddon" (Thomas, Jones et al. 2019). However, it seems the overall trend is similar to the vertebrates' – some species are declining and some are increasing in abundance (Dornelas and Daskalova 2020, Klink, Bowler et al. 2020, Wagner, Fox et al. 2021); but the underlying data are even less complete and more biased than for vertebrates (Leather 2018, Cardoso and Leather 2019, Thomas, Jones et al. 2019). All in all, the contrasting reactions to the same data and principally similar results show the effect of values and normative views. Most biologists view extinctions as negative phenomena, but while for some the current rates of loss appear catastrophic, for others those are unfortunate yet not too tragic side-effects of human well-being. While there is no way to resolve such value-disputes, it might be more reasonable and robust to approach the issue of global biodiversity change from a more utilitarian perspective, focusing on the effect on ecosystem functioning and resilience.

So far, I have claimed that the state of biodiversity is changing, and I outlined some trends, but what is it that causes the change? The most important direct drivers of anthropogenic change are usually identified as land and sea use change, exploitation of organisms, climate change, pollution, and invasive and alien species (Maxwell, Fuller et al. 2016, Díaz, Settele et al. 2019). While there are multiple sub-categories (deforestation, agricultural expansion, urban expansion) and names (habitat loss, habitat degradation, etc.) for land use change, as a group it is usually identified as the primary driver of biodiversity change (Tilman, Clark et al. 2017). Although already for the past several thousands of years there were arguably few places on Earth completely unaffected by human presence (Ellis, Gauthier et al. 2021), the current rate and quality of change is much more pervasive (Winkler, Fuchs et al. 2021), and is expected to remain so well into the future (Newbold, Hudson et al. 2015, Powers and Jetz 2019). Nevertheless, the rate and direction of change is not homogenous around the globe – for example, in the global South the forested area is decreasing while in the global North it is increasing. (Winkler, Fuchs et al. 2021). Furthermore, it is not only the overall extent of viable habitat (e.g., forest, grassland) - as inferred from the species-area relationship –, but the quality of the remaining habitat is important as well (Watson, Evans et al. 2018, Dullinger, Essl et al. 2021). Ecosystem decay caused by habitat degradation (e.g., selective logging, edge effect, or wildfires, but also intensification of agriculture) further exacerbates the biodiversity loss (Barlow, Lennox et al. 2016, Chase, Blowes et al. 2020), which also stands for protected areas of which one third is still intensely affected by human pressures (Jones, Venter et al. 2018). In fact, the current biodiversity loss could perhaps be better explained by the "*displacement of species-rich cultural natures sustained by past societies than the recent conversion and use of uninhabited wildlands*" (Ellis, Gauthier et al. 2021). Also there might be thresholds in the amount of habitat below which the biodiversity response is more abrupt (Boesing, Nichols et al. 2018). There is a general agreement that the total amount of habitat largely determines biodiversity, yet - although in conservation practice it is widely accepted that a single large area is preferable to several small patches - the effect of habitat fragmentation *per se* is still debated. The preference for large contiguous areas stems from the assumption that fragmentation itself causes additional biodiversity loss, but lately some scientists claim there is no empirical proof of a general negative effect of fragmentation on a landscape level (Fahrig 2017, Fahrig, Arroyo-Rodríguez et al. 2019, Watling, Arroyo-Rodríguez et al. 2020). Other authors claim such conclusions are problematic and biased (Fletcher, Didham et al. 2018), and in the case of high degree of fragmentation the effect is negative and additional to habitat loss (Hanski, Zurita et al. 2013, Hanski 2015, Rybicki, Abrego et al. 2020). Nonetheless, although the exact relationship is non-trivial, it seems clear that as more land is converted for an intense human use, less is available for wild organisms, so less biodiversity remains.

Throughout the history, overexploitation of organisms has competed with habitat change for the infamous position of the most injurious driver. Already since the end of the last ice age, when excessive hunting led to the extinction of (among many others) most *Elephantidae* species, people have driven to the brink of extinction many animal species, and the practice of overhunting is continuing until today, for example in the tropics (Benítez-López, Santini et al. 2019). Nowadays, even seemingly intact forests could be largely devoid of animals (Benítez-López, Santini et al. 2019), and poaching affects even such taxa as primates (Estrada, Garber et al. 2017). Next to subsistence hunting, currently even more pervasive phenomenon is the wildlife trade (Morton, Scheffers et al. 2021). Another widely discussed driver are invasive species (Van Kleunen, Dawson et al. 2015, Pyšek, Hulme et al. 2020). Which, on the one hand, cause declines in population of native species, on the other hand the novel species could substitute ecological functions in the changing environmental and climatic conditions. Changing climate is another significant driver of biodiversity change (Steinbauer, Grytnes et al. 2018), and it is expected to become even stronger in the future (Trisos, Merow et al. 2020). With the changing climate, species are forced to shift their ranges, change relative abundance within the species ranges, and also to subtler changes in the timing of activity and microhabitat use (Pecl, Araújo et al. 2017). Ultimately, "*as climate changes, species must either tolerate the change, move, adapt, or face extinction*" (Pecl, Araújo et al. 2017). Interestingly, so far there has been a radically different scale of response to climate change between marine and terrestrial ecosystems (Antão, Bates et al. 2020). Changes in biosphere may feed-back the climate change (Bonan and Doney 2018), leading to ever larger impacts on biodiversity. On the other hand, the currently more diverse ecosystems could be more resistant to the changing climate (Isbell, Craven et al. 2015). This indicates that drivers of change rarely come in isolation (Williams, Freeman et al. 2021), and "*species rarely experience identical impacts of environmental change due to interactions between threats, landscape composition, and the scale at which they experience environmental drivers*" (Oliver, Heard et al. 2015). Furthermore, the response to the same pressure might differ even between individuals of the same species (Merrick and Koprowski 2017). Among the most crucial interactions for the future state of biodiversity is the one between land use and climate, as the habitat composition can either mitigate or exacerbate the impact of climate change (Williams and Newbold 2020, Williams, Freeman et al. 2021, Williams and Newbold 2021). Next to that, land use change interacts with wildlife trade

(Symes, Edwards et al. 2018), and invasive species establish better in degraded habitats (Early, Bradley et al. 2016). Despite the fact we tend to treat those drivers as isolated phenomena, as far as they are anthropogenic, all are in a strong correlation with the growth of the Gross Domestic Product (GDP) (Otero, Farrell et al. 2020) and the size of human population (Crist, Mora et al. 2017). Although the overall trend of global biodiversity might not be clear (as discussed above), the effect of current drivers might not have manifested yet, but might be present in the form of "mass extinction debts" (Sage 2020).

To map the current state of biodiversity, predict future states, and effectively manage world's ecosystems we not only need to directly measure the impact of those drivers as it manifests itself (which is very costly and laborious), but also need to be able to model and predict the state of biodiversity based on some proxy variables. There are many community ecology models - theoretical and pragmatic - that can help predict the effect of changes in community (e.g., after a loss of an overhunted predator) (Levine, Bascompte et al. 2017) or changes in the environment (spatial ecosystem models were nicely reviewed by DeAngelis and Yurek (2017)). Nevertheless, "*predictions under global change or following biological invasions might differ markedly depending on the approaches applied*" (Mateo, Mokany et al. 2017). There are many macroecological theories and hypotheses on what determines the number of species in a community, and whether the number of species in a community is environmentally limited. It seems likely that while communities are unsaturated, they are still constrained in their composition by various drivers (Mateo, Mokany et al. 2017). This could indicate to a theory of biodiversity which predicts that there is, simply said, an equilibrium determined by a concurrence of environmental (area, resource availability etc.) and ecological (speciation, extinction, species traits etc.) drivers (Storch and Okie 2019, Storch, Šímová et al. 2021). Models based on such theory could be used to accurately predict changes in biodiversity resulting from anthropogenic environmental change, granted all the underlying processes and mechanisms are accurately described, which is, if possible, still a task for future research. Withal, it is clear that such ecological-theory models are not yet operationalizable in the context of studies of interactions in the global social-ecological system, as this work strives to be. In the case such studies apply an ecological model, it is usually based on species-area relationship (SAR) (Harte, Smith et al. 2009) to describe the effect of land use change.

As shown above, people affect the environment around them and hence the biodiversity, yet we are also entirely dependent on the services and contributions nature provides us, hence the more utilitarian approach proposed above. This link between society and nature (although such distinction is problematic, it is useful in this context) is embraced in the concept of ecosystem services (ES) (Costanza, d'Arge et al. 1997, Millennium Ecosystem Assessment 2005), which has been recently re-conceptualized as nature's contribution to people, following the critique that ES is based predominantly on the stock-and-flow framing and that it does not take into account insights and tools developed in social sciences and humanities (Díaz, Pascual et al. 2018). In this concept there are recognized material, nonmaterial, and regulating contributions, while culture permeates all categories (Díaz, Pascual et al. 2018). Nevertheless, the provisioning of those services or contributions is largely dependent on functioning of ecosystems, which to a large degree depends on biodiversity. Single ecosystem functions – even biomass production (Liang, Crowther et al. 2016, Emmett Duffy, Godwin et al. 2017) – and higher multifunctionality are commonly associated with higher biodiversity (Lefcheck, Byrnes et al. 2015), although the relationship is variable (van der Plas 2019), scale-dependent (Gonzalez, Germain et al. 2020), and there are still many limitations to biodiversity-(multi)functionality relationship assessments (Manning, Van Der Plas et al. 2018). It was also indicated that some ecosystem functions are probably not dependent on species richness but more on the abundance of common species (Winfree, Fox et al. 2015), or that trait diversity is a better predictor of functionality than taxonomic diversity (Mouillot, Graham et al. 2013). There are generally three mechanisms by which biodiversity loss affects ecosystem functioning: with additive mechanism, ecosystem functioning decreases linearly with species loss; there is higher than proportional loss of functionality as species with keystone interactions disappear; but lower than proportional functionality loss with disappearance of redundant interactions (Valiente-Banuet, Aizen et al. 2015). On the other hand, biodiversity effect on ecosystem services is possibly undervalued as there is usually not included the insurance effect of biodiversity (Isbell, Gonzalez et al. 2017). Thus, while not many species might be necessary for the current functionality, higher biodiversity levels might be needed to provide similar function in an altered environment (Oliver, Heard et al. 2015). In consequence, "*conservation efforts might be short-sighted if they prioritize the current crucial component of biodiversity without considering whether the same set of components will remain important in the future*" (Isbell, Gonzalez et al. 2017). Furthermore, biodiversity could be considered a valuable ecosystem function or service by itself (Garland, Banerjee et al. 2021), since contact with nature has a positive impact on human health and overall well-being (Sandifer, Sutton-Grier et al. 2015).

2.3 Biodiversity footprint

The primary goal of this study is to quantify "biodiversity footprint" coupled with international trade of the Czech Republic. In a broader sense, this means to analyze and quantify the effect and impact the products imported to the Czech Republic have on the state of biodiversity in the country of origin. "Footprint" is another one of the words frequently used in environmental research and policy. Yet again, it has no precise meaning, and there is little consensus on what should footprint study look like and comprise of (Matuštík and Kočí 2021). The term "footprint" was first used in this context by Rees (1992), who designed the indicator called ecological footprint, and thus established the field. The original purpose was to create a comprehensive sustainability indicator, to measure "*the aggregate area of land and water in various ecological categories that is claimed by participants in that economy to produce all resources they consume, and to absorb all their wastes they generate on a continuous basis, using prevailing technology*" (Wackernagel and Rees 1997). Nevertheless, the indicator is not quite so comprehensive. The ecological footprint became very popular, though, among other reasons, thanks to outputs like National Footprint Assessment (Lin, Hanscom et al. 2018) and Overshoot day; and it initiated a boom of footprint research. Carbon footprint, describing the impact on the climate, and water footprint followed suit in the early 2000s. Other environmental problems got their footprint indicators, for example, material footprint, nitrogen footprint, phosphorus footprint, and there were also proposed social footprints (Čuček, Klemeš et al. 2012). Although not always, footprints usually take the consumption perspective, meaning that the responsibility for the environmental impact is allocated to the final consumer – as opposed to the production perspective, where the responsibility is allocated to the country in whose territory the impact takes place. Following the criticism that problem-shifting might occur when only solitary footprints are employed, it was proposed that a full Footprint family should be assessed (Galli, Weinzettel et al. 2013, Fang, Heijungs et al. 2014).

Although footprints grew to be very popular, there are also several problems with the concept. While a part of the community sees footprints as resource-use and emissions oriented (Vanham, Leip et al. 2019), others argue that footprints should indicate a harmonized impact, as it is done in Life Cycle Assessment (LCA) (Pfister, Boulay et al. 2017). Next to the overall identity problem, there are specific issues for each of the footprints. Since its establishment, ecological footprint has been heavily criticized for its many methodological shortcomings (Giampietro and Saltelli 2014, Patterson, McDonald et al. 2017). Although ecological footprint inaugurated the field, it might be time it is abandoned, as it is a composite indicator, which nevertheless provides no additional information to other footprints (Matuštík and Kočí 2021). The problem with carbon footprint is that there are many approaches to quantification and also different definitions of what carbon footprint is (Wright, Kemp et al. 2011). This may lead to contrasting results for the same objects of analysis (Heinonen, Ottelin et al. 2020). Nevertheless, now there are several standards for carbon footprinting (e.g., ISO 14067 and ISO 14064), largely based on LCA methodology, that aim to put order in the chaos. Water footprint is the best example of the dichotomy between the pressure-based approach, as used in the original method by Hoekstra (2003, 2017), and the LCA-based impact approach (Boulay, Bare et al. 2018). Similarly as there is not the same environmental impact when a m^3 of water is withdrawn in a desert or in a rainforest, the environmental and socio-economic impacts of the use of, e.g., a tonne of sand is different than when a tonne of gold is used, which are, nevertheless, usually summed to form a composite material footprint (Fang and Heijungs 2014). Overall, the common characteristic of the less-conventional footprints is that the methodology is still evolving, and thus rarely any two indicators with the same name use the same methodology. Much has been written about this elsewhere (Matuštík and Kočí 2021) though, and I do not want to dwell on it too much. All in all, it can still be said that footprints, understood in the broadest sense as "indicators of human pressure on the environment" (Hoekstra and Wiedmann 2014), are a very popular concept that can provide useful information on the relations in the social-ecological system (Wiedmann and Lenzen 2018).

The footprints most relevant to this study are land footprint and biodiversity footprint. A measurement of consumption of land, land footprint, has already been a crucial part of the original ecological footprint indicator, though. In fact, the ecological footprint (Rees and Wackernagel 1996) could also be called land footprint, because the unit of calculation is a global hectare of land, and the name itself is closely connected to land - the area of land we leave our imprint on. While there are several types of appropriated land accounted in the ecological footprint: land as the source of food and timber (productive land), the land appropriated for infrastructure and housing (build-up land), the hypothetical forested land needed to absorb carbon dioxide emission, and also the area of appropriated water (Lin, Hanscom et al. 2019); conventional land footprint is limited to the first and sometimes the second part, never the carbon segment. Many assessment techniques, like Material Flow Analysis (MFA), Multi-regional Input-Output (MRIO), or LCA, with further varying impact assessment methodologies are used to quantify a land footprint (O'Brien, Schütz et al. 2015, Perminova, Sirina et al. 2016). In the purely pressure-based approach, all kinds of land can be summed to the total occupied-land area, disregarding the differences in socio-economic or ecological value. Some land footprint studies employ a similar methodology to the ecological footprint accounting, where the appropriated land area is expressed in the unit of an hypothetical hectare of global average productivity (Weinzettel, Hertwich et al. 2013). Three basic types of land use can be evaluated in the productive-land footprint - cropland, grassland, and forestland. Nevertheless, people do not consume the land directly (unlike build-up land), but only the crops, vegetables, meat, and timber harvested from the land. This brings an obvious challenge how to translate the amount of consumed product to the area of land. While nation-specific yield factors are usually used, it is clear that even in the smallest countries the land productivity (soil fertility, topography, climatic conditions etc.) is variable (Schaffartzik, Haberl et al. 2015). Indeed, "*products derived from lower productivity cropland may have a higher cropland footprint than products derived from a higher productivity cropland, but this does not necessarily mean that they are less sustainable or that they are contributing in a greater way to cropland scarcity*" (Ridoutt and Navarro Garcia 2020). Notwithstanding that although organic and sustainable farming practices usually present lower yields than intensive industrial agriculture, which would result in a higher land footprint, such practices are usually less damaging to the environment. To deal with this, the appropriated land area can be translated to a land-use risk index based on nutrient balance (Taherzadeh, Bithell et al. 2021). In the Environmental Footprint LCA methodology by European Union, the land-use is expressed as a Soil Quality Index measuring the impact on erosion, water infiltration etc. (De Laurentiis, Secchi et al. 2019). Another approach to land area harmonization can be through the potential net primary productivity of the land (Weinzettel, Vačkářů et al. 2019). Such approaches shift the focus of the land footprint indicator from the direct effect on the production of food and resources to environmental issues. All in all, the various methodologies of land footprint have been applied to various study objects. Land footprint has already been quantified for many products, e.g., soybeans (Liu, Yu et al. 2021), cacao (Armengot, Beltrán et al. 2021), but also solar power (Wu, Shao et al. 2021). It was used to assess the domestic and distant land footprint of many countries, e.g., Denmark (Osei-Owusu, Kastner et al. 2019) or the United Kingdom (de Ruiter, Macdiarmid et al. 2017); single continents (Steen-Olsen, Weinzettel et al. 2012) and the entire global trade were analyzed (Weinzettel, Vačkářů et al. 2019). It was also applied to analyze specific economic sectors, mainly the food sector with changing dietary trends (Rizvi, Pagnutti et al. 2018) or the emerging bioeconomy (Liobikiene, Chen et al. 2020).

As described above (Section 2.2), multiple anthropogenic drivers affect the state of natural ecosystems and biodiversity. Although this may change with the growing severity of the climate crisis, land use and land cover change are arguably the major drivers of biodiversity loss. Accordingly, the biodiversity footprint indicator is usually focused on tracking the effect of land use, in most cases as the only driver that is considered. Indeed, biodiversity footprint could often be seen as a further extension to the land footprint, shifting the focus from land area as a resource to the impact land use has on ecosystems. Still, other drivers of biodiversity loss were included by some authors as well. Although there have been studies linking agriculture and international trade to biodiversity impacts already before (e.g., (Donald 2004) or (Naidoo and Adamowicz 2001)), the field of biodiversity footprinting *per se* has only about a decade-long history, which is even shorter than most other footprints. Hence, the variety of approaches to biodiversity footprint calculation is very broad. Biodiversity footprint was probably first calculated in a study by Lenzen, Moran et al. (2012). They linked the production of "implicated commodities" to the threat causes for threatened species based on IUCN and Bird Life International data; and they analyzed global trade in those commodities, comparing imports and exports of biodiversity threats (Lenzen, Moran et al. 2012). This study was one of the first attempts to evaluate the biodiversity threats linked to global trade and hence it provided valuable and novel insights. Yet there are several limitations to the impact quantification methodology. First, the link between the production sector and biodiversity here is not causal and quantitative, but correlative in a binary threat/no-threat manner; and the link of the threats to the monetary value of consumption is not exactly clear. Second, the effect on biodiversity is here limited only to the threats to animal species classified as vulnerable, endangered, and critically endangered according to IUCN (*iucnredlist.com*), which represents only a small fragment of the picture of global biodiversity. Such approach already predetermines the final conclusions that northern "developed" countries are net-importers of biodiversity threats, because there are usually many more species in tropical "developing" countries and thus more threats are likely. A similar methodology was later used to evaluate the suitability of MRIO analysis to calculate biodiversity footprints (Moran, Petersone et al. 2016), and the same method was also used to identify "species threat hotspots from global supply chains" (Moran and Kanemoto 2017).

Later in the year of 2012, Hanafiah, Hendriks et al. (2012) took a completely different approach to biodiversity footprint calculation. They built on the methodology of ecological footprint accounting, but instead of land productivity they factored in the Mean Species Abundance

(MSA), i.e., "*the remaining mean species abundance of original species, relative to their abundance in pristine or primary vegetation, which are assumed to be not disturbed by human activities for a prolonged period"* based on (Alkemade, van Oorschot et al. 2009). They multiplied the land occupation values with factors of MSA loss (i.e., 1 - MSA) respective to the land use type, using MSA factors from the GLOBIO3 model (Alkemade, van Oorschot et al. 2009). Next to this, they also derived MSA factors of loss related to climate change caused by CO_2 emissions; although considering the knowledge and data deficiency described above (Section 2.2), the accuracy of those values is likely very limited. Overall, they interpret this biodiversity footprint as the "*global area that is required to compensate for the mean species abundance loss caused by direct land use [and fossil-fuel based CO_2 emissions] for the life-cycle of a product*" (Hanafiah, Hendriks et al. 2012). This is in line with the logic of the ecological footprint and its narrative of "we need more than one planet to supply the resources we consume." Applied to species abundance it does not seem to make sense though. The species abundance is thus seen as another product of land that could be produced elsewhere, not a proxy to the state of the ecosystem. Although the indicator is meant as a purely abstract warning on the unsustainability of human consumption, it is clear that such thing as compensation of the lost species abundance elsewhere is not really possible, definitely not in such a straightforward manner. Withal, they used this method to evaluate several product categories and compared the results to ecological footprint. While they were among the first to call this the biodiversity footprint, similar "impact characterization" approaches were taken already before in the field of LCA (Elshout, van Zelm et al. 2014, Souza, Teixeira et al. 2015).

Indeed, LCA practitioners attempted to devise methodologies to link land use with its impact on biodiversity already in the early 2000s, which led to formation of a working group under UNEP-SETAC resulting in a guideline document (Koellner, de Baan et al. 2013). Their notions were considered in the development of the method to assess the impact on relative species richness by de Baan, Alkemade et al. (2013) that was later used in the Impact World+ methodology (Bulle, Margni et al. 2019). The guideline was also used to create the LC-Impact methodology (Verones, Hellweg et al. 2020). Here they derive the link from land use transformation and occupation to a resulting "global fraction of potentially disappeared species" (Chaudhary, Verones et al. 2016), mainly based on countryside species-area relationships weighted with "vulnerability scores" (Chaudhary, Verones et al. 2015). This impact assessment method was applied to analyze international food trade, yet without being labelled as LCA or footprint (Chaudhary and Kastner 2016). Later it was further updated and refined (Chaudhary and Brooks 2018), and, interestingly, it started to be called "biodiversity footprint" as well as life cycle assessment. Chaudhary and Brooks (2019) used a similar SAR-based biodiversity-impact assessment method to analyze impact of national consumption and global trade, now without a single mention of LCA and only a solitary use of the term footprint. Finally, Koslowski, Moran et al. (2020) use LC-Impact and ReCiPe (Huijbregts, Steinmann et al. 2017) methodologies, developed for LCA, to calculate biodiversity footprints of Europe and to analyze the role of urbanization and income (their terminology), and Bjelle, Kuipers et al. (2021) used LC-Impact to analyze if there is any decoupling of biodiversity footprint from income. This is a nice example of the methodological and terminological chaos in the footprint field, where even the same authors using almost similar methodology call the method of analysis sometimes a footprint, sometimes an LCA, sometimes both, sometimes neither. In the end, one approach to identify a footprint study is that "footprint is what is footprint called" (Matuštík and Kočí 2021). Another, which I'll further apply here, is the approach used by Marques, Verones et al. (2017) where biodiversity footprint is what seems like a biodiversity footprint, i.e., a metric that captures "*the direct effects [on biodiversity] of an activity as well as the indirect effects that are transferred along a supply chain*." Such metric can be calculated through both LCA and MRIO, as reviewed by Marques, Verones et al. (2017).

From the previous paragraphs it is apparent that multiple biodiversity variables were used to derive a biodiversity footprint. As already presented, there is the "number of threats" used by Lenzen, Moran et al. (2012), the loss of Mean Species Abundance used by Hanafiah, Hendriks et al. (2012), and the Potentially Disappeared Fraction (PDF) of species commonly used in LCA (Marques, Robuchon et al. 2021). Another aggregate indicator is the Biodiversity Intactness Index (BII) used for a related analysis by Newbold, Hudson et al. (2015). Each of those indicators provides different information and thus different footprint results (Marquardt, Guindon et al. 2019). Biodiversity impacts were also expressed as the number of bird species potentially lost (Marques, Martins et al. 2019), or a "mean biodiversity richness of taxa" (McManamay, Vernon et al. 2021). While most studies aim to include all land use types, some studies are limited to only some. For example Sandström, Kauppi et al. (2017) calculate biodiversity footprint coupled with food production, Ridoutt and Navarro Garcia (2020) focus on cropland, and next to biodiversity impacts they assess cropland scarcity and potential malnutrition. The scope of the studies varies greatly as well, from studies focused on individuals or households (Koslowski, Moran et al. 2020), products and product categories (Asselin, Rabaud et al. 2020), to biodiversity footprints of nations, e.g. Finland (Sandström, Kauppi et al. 2017). Nevertheless, the most common approach is to analyze the flows in the global trade. Although all studies take the "consumer perspective," a critical difference between those approaches is that while the studies focused on individuals or households quantify the total footprint of the subject, the studies focused on global trade flows usually account only extraterritorial, not domestic footprints. The studies also differ in whether they analyze the footprints in a single year, as a snapshot picture, or whether they analyze longer-term trends, like Bjelle, Kuipers et al. (2021). Biodiversity footprints were also calculated for predicted future land use change scenarios (Chaudhary and Mooers 2018) or the land use under the Shared Socioeconomic Pathway scenarios (Marquardt, Doelman et al. 2021).

Considering the variety in the approaches to biodiversity footprint quantification, it is clear that, unlike ecological footprint or water and carbon footprint to a degree, it cannot be considered an established indicator with consistent methodology and comparable results. So far, the term "biodiversity footprint" is almost haphazardly used as a synonym to "biodiversity impact" or "biodiversity damage." This means, on the one hand, that results of different studies are barely comparable. On the other hand, it also means that the methodology is still very much open to modifications and improvements.

3 Materials and methods

As mentioned previously, and as the name of this thesis suggests, the main goal of this work is to quantify the biodiversity footprint of Czech consumption. As described above (Section 2.3), however, biodiversity footprint cannot yet be seen as an established indicator with a standard quantification methodology. Using the name biodiversity footprint says little about the methodology of a study. In this thesis, assessing a biodiversity footprint means to quantify the impact on biodiversity coupled with international trade in products that are consumed in the Czech Republic but produced elsewhere. While the object of interest here is the Czech consumption, the scope of impact quantification is global, i.e., the entire foreign footprint, as far as possible. This study is retrospective, i.e., the patterns of trade in the past are analyzed; no direct predictions on what the impact will be in the future are made. The main research goal would be served by answering several research questions, like: *What is the total biodiversity footprint of trade of the Czech Republic? Which countries and regions are the most affected by the Czech consumption? Consumption of which products or product categories is the most culpable for this biodiversity impact? How did the biodiversity footprint evolve in time? How does it compare to the impacts exported from the Czech Republic?*

The above listed research questions are to be answered by the results. Nevertheless, to reach meaningful results, several methodological and data questions need a solution as well. *Which products are traded with the Czech Republic, and what are their countries of origin?* Section 3.1 describes the trade-data used and how were they processed. *What land use is coupled with production of those products?* Section 3.2 describes how land occupation was quantified. *What impact on biodiversity is coupled with this land use?* Section 3.3 describes the biodiversity data used for impact characterization and the indicator Biodiversity Intactness Index. Furthermore, Section 3.4 describes the specific methodology of biodiversity impact assessment and the rationale for this approach.

3.1 Trade

Production of some goods (especially products of agriculture and forestry) is usually limited to specific regions of the world, and the specific production processes and potential environmental impacts they cause often differ between those regions. Nevertheless, consumption of those products in the globalized economy is not bound by boundaries of countries or regions. There are multiple approaches to link consumption and production (Tukker, Giljum et al. 2018). It can be made through bottom-up methods like LCA, where production tree is constructed for each product and then evaluated. Nevertheless, when patterns of trade are analyzed, top-down methods are usually used. The most common such technique is Multi-Regional Input-Output analysis, which will be used here as well. Input-output models track flows between economic sectors of a country in a certain year, multi-regional databases cover multiple regions and countries. For footprint analyses, an economic input-output matrix that usually shows monetary flows is coupled with environmental data (e.g., CO_2 emissions, water use), this is called Environmentally-Extended Input-Output analysis (EE-IO, or EE-MRIO for multi-regional input-output) (Kitzes 2013).

The basic framework of Input-Output analysis itself was created already in the 1930s by Leontief and was developed since (Miller and Blair 2009). The increasing complexity and fragmentation of global supply chains means, though, that input-output tables measuring only bilateral trade between countries (such as the OECD trade database) are insufficient to account for all embodied environmental impacts; and hence a considerable effort has been made to develop global MRIO databases (Tukker and Dietzenbacher 2013). An ideal global MRIO database would be "*as detailed as possible in terms of sectors and products, with a set of socio-economic and environmental extensions as extensive as possible, covering the globe and discerning as many as possible countries and regions, including long time series*" (Tukker and

Dietzenbacher 2013). Several more or less successful attempts were made to create such a database. The Global Trade Analysis Project (*gtap.agecon.purdue.edu*) have published aggregate input-output data since the 1990s, and there is already version 10 of the GTAP database (Aguiar, Chepeliev et al. 2019). Another well-known MRIO databases are the Eora database (Lenzen, Moran et al. 2013), the ICIO (Yamano and Webb 2018), or the WIOD (Dietzenbacher, Los et al. 2013), and there are others like those by Bruckner, Giljum et al. (2012) or Andrew and Peters (2013). Finally, there is EXIOBASE (Wood, Stadler et al. 2015), updated version of which is used in this study. The databases differ in their time coverage, sector detail, and country aggregation (Marques, Verones et al. 2017). I chose EXIOBASE because it seems to best fit the scope of this study with its coverage.

The first version of EXIOBASE was published in 2015 (Wood, Stadler et al. 2015) and it was upgraded to EXIOBASE 3 in 2018 (Stadler, Wood et al. 2018). Compared to the first two versions, which provided data for only single years (2000 and 2007, respectively), EXIOBASE 3 provides a time series from 1995 to 2011 (Stadler, Wood et al. 2018). The main strength of this database is that it provides high-level of consistent sectoral coverage (200 products and 163 industries for all covered countries), which is considerably more than most other databases, except for Eora. In Eora the sectoral coverage is higher, but it is inconsistent among countries, so it does not allow direct comparison. One drawback of EXIOBASE 3 is that it provides data only for 44 countries, while the rest is aggregated into five Rest of the World (RoW) regions. Although those countries account for almost 90% of global gross domestic product (Stadler, Wood et al. 2018), over one third of the world population and land area is thus aggregated in the RoW regions (Bjelle, Többen et al. 2020). This is resolved in the latest version of the database, EXIOBASE 3rx (Bjelle, Stadler et al. 2019), where the data are disaggregated to comprise 214 countries. The precise process of building the database is described in (Stadler, Wood et al. 2018) and in (Bjelle, Többen et al. 2020). The database provides a time series from 1995 to 2015 and an environmental extension of land occupation. The database is open-licensed and free to be downloaded from *doi.org/10.5281/zenodo.2654460* in *.mat* format. I downloaded the data in November 2021. The data were processed in MATLAB R2021b software, the data on imports and exports for the Czech Republic in each year were extracted from the trade cube and further processed to be used to calculate the land occupation embodied in the trade, which will be described in the next section.

3.2 Land use

The data on land occupation embodied in international trade were also derived from the EXIOBASE 3rx database (Bjelle, Stadler et al. 2019). To develop the land use extension, the authors harmonized, following a closed budget approach (Erb, Gaube et al. 2007), several land use and land cover datasets - among others, the data from the HYDE project (Klein Goldewijk, Beusen et al. 2017), data by European Space Agency (*http://maps.elie.ucl.ac.be/CCI/viewer/*), or the Human Footprint project (Venter, Sanderson et al. 2016) – to create a spatially explicit land cover maps. Then they used statistics by FAOSTAT to disaggregate the data to match the sectors in EXIOBASE and derive the impact intensity values. The exact procedure is described in the Additional file 1 of (Bjelle, Többen et al. 2020). The extension is an integral part of the EXIOBASE 3rx download. Using MATLAB software, I multiplied the monetary values of trade by the factors of impact intensity per monetary unit specific for each country and product. Thus, the occupied land area embodied in the consumption of each product is derived. The occupied land area is divided between four land use categories, i.e., cropland, forest, pasture, and other. For the twenty-one years available in the database, this results in 84 tables of embodied land use in the products imported to the Czech Republic, plus 84 tables describing export form the Czech Republic in those years.

3.3 Biodiversity Intactness Index

While the previous two steps comprised mainly of processing database data, i.e., processing data other researchers collected, the phase of biodiversity impact characterization, as described in this section, is unique for this study. The indicator of biodiversity, or let's say the indicator of the state of ecosystems, selected for this study is the Biodiversity Intactness Index (Scholes and Biggs 2005). In the original definition "*the BII gives the average richness- and area-weighted impact of a set of activities on the populations of a given group of organisms in a specific area*" (Scholes and Biggs 2005). In simple words, this index describes the state of biodiversity (ecosystem) by comparing the current state to an "undisturbed" reference state. Of course, as any biodiversity indicator, it is far from perfect. Some authors, for example, criticize that BII insufficiently captures the "variation" aspect of biodiversity (Faith, Ferrier et al. 2008), but as discussed above (Section 2.2) there are many elements of biodiversity, and one indicator can never capture them all. One obvious problem of BII, though, is how to set the reference state. Ideally, the reference should be the state of the ecosystem before a human conversion, e.g., the primary forest present before the conversion to plantation, or the grassland before the conversion to cropland. Since many of the land cover conversions happened hundreds of years ago, we can only guess how exactly the ecosystem looked before. The common solution is the so-called space-for-time substitution, where the state of the original ecosystem is approximated by some purely natural ecosystem undisturbed by humans (an ecosystem where BII=1) nearby. The fact that even most land classified as wilderness has been somehow affected by humans as well may result in overestimation of the BII, a pattern pointed out by Martin, Green et al. (2019). Another obvious problem, common to all global biodiversity metrics, is the lack of data (Section 2.2). Nevertheless, while the accuracy of any current global BII estimations is still limited, this does not disqualify the indicator itself (Newbold, Sanchez-Ortiz et al. 2019); and BII was embraced by multiple authors and organizations, for example, (Marques, Robuchon et al. 2021), (Steffen, Richardson et al. 2015), or IPBES (*ipbes.net/core-indicators-0*). Nonetheless, gathering global biodiversity data is the major hurdle for any biodiversity footprint assessment. Thankfully, now there are databases that gather global data about biodiversity like BioTIME (*biotime.st-andrews.ac.uk*) or PREDICTS (*predicts.org.uk*). The PREDICTS database collates primary data on species abundance and occurrence from hundreds of studies (Purvis, Newbold et al. 2018). The data from the PREDICTS database were, for example, used to assess the effect of land use on several biodiversity variables (Newbold, Hudson et al. 2015). While the early studies of BII (Scholes and Biggs 2005) relied heavily on expert opinion, and were criticized for it (Rouget, Cowling et al. 2006), such global databases allow to create global biodiversity intactness maps based on observational data (Newbold, Hudson et al. 2016). Here, I use the latest available global BII map constructed from the PREDICTS data (Sanchez-Ortiz, Newbold et al. 2019).

Following and further refining the approach used by Newbold, Hudson et al. (2016), Sanchez-Ortiz, Gonzalez et al. (2019) developed a global statistical model of BII describing the relationship between the available species-abundance and species-richness data with multiple predictor variables like land use, land use intensity, distance to roads, or human population density. The detailed description of the modelling procedure and the data used can be found in (Sanchez-Ortiz, Gonzalez et al. 2019). They used the original abundance-based approach by Scholes and Biggs (2005) to calculate BIIs from the model, but they also derived BIIs using species richness to address the critique that abundance-based BII may overlook losses as they might be compensated by increases in common species (Sanchez-Ortiz, Gonzalez et al. 2019). Nevertheless, the exact calculation of BII here is slightly different than the equation by Scholes and Biggs (2005) which relied on expert-derived intactness coefficients. Here they compared the modelled biodiversity to the "*expected species richness and total abundance for a grid cell* *composed entirely of primary vegetation with minimal human use*," as described in the supplementary document to (Newbold, Hudson et al. 2016). Evaluating the two types of BII provides additional information as "*the use of total abundance might be more appropriate when analyzing the amount of ecosystem service provision. However, richness-based metrics give equal weight to rare and common species, which might be more relevant for stability of ecosystem service provision.*" (Sanchez-Ortiz, Gonzalez et al. 2019). The global maps of abundance- and richness-based BII are openly available for download (*doi.org/10.6084/m9.figshare.7951415*) as raster (*.tiff*) with resolution of 30 arc sec (approx. 1 km^2) in WGS 1984 projection.

3.4 Biodiversity impact characterization

To derive the country- and land-use-specific BII characterization factors for land occupation, I combined the information from the BII map with the global map of dominant land cover by FAO (2014). The map I used depicts the dominant land cover type for each grid cell with 30 arc sec resolution (i.e., the same resolution as the BII map) in WGS 1984 projection, and it is openly available as GIS raster layer. To process the spatial data, I used ArcGIS Pro 2.9.1 software by Esri. First, I converted the raster layers to point layers which I joined spatially. Thus, each grid cell of 1 km^2 (each point that represents the cell) was assigned a BII value and a land cover type. Furthermore, I assigned each cell a country identifier using the layer *World Country Boundaries 2019* by Esri. I assume the traded products are not produced in primary ecosystems which have high BII. To limit the effect of those ecosystems on the characterization factors, I erased the cells located within protected areas using the *WDPA - World Database of Protected Areas* layer (UNEP-WCMC and IUCN 2021) - as the best available approximation of natural ecosystems where no, or only limited, production of the traded products takes place. The final characterization factors depict the median value of biodiversity intactness in each country for the four land use categories used in EXIOBASE. Since the land use classification nomenclature slightly differs between EXIOBASE and the FAO map, I used the following pairing (the first is the category in EXIOBASE, the second the category in the FAO map): cropland = cropland; forest = tree covered area; pasture = grassland; other = artificial surfaces + shrubs covered area + sparse vegetation. I chose to calculate median values of BII, rather than mean, to limit the effect of outliers, some cells with unusually large or low BII values or extreme values potentially caused by some misalignment between the two maps. Obviously, throughout a country, especially in large countries, the state of biodiversity might differ even between, for example, fields where the same crop is grown in the same way, let alone between organic and industrial production. Similarly, even biodiversity in managed forests largely differs depending on the level of management intensity, harvesting systems etc. While accounting for those differences would lower the uncertainty of biodiversity footprint results, the benefit of such differentiation is currently limited by the consumption and trade data aggregated at the level of land use type, product sector, and country. Therefore, the median BII represents the value most likely associated with the land use type in the country. This is the highest level of resolution the available data currently allow. Furthermore, those characterization factors represent the BII associated with a landscape where the respective land use is dominant, rather than BII of that land use specifically. This adds further variability to the BII values since the fraction of the land-use in the said grid cell is variable. On the other hand, I believe such "landscape approach" captures the overall effect on biodiversity better than the BII if only pure land-use categories were reflected, especially considering that biodiversity on a plot can hardly be separated from the surrounding landscape. Nevertheless, such debates are purely academical because the accuracy and uncertainty of the available biodiversity data absolutely do not allow to demonstrate any difference.

Several approaches are possible to how to apply these biodiversity-impact characterization factors on the pressure data (land occupation embodied in consumption). Different approaches result in different behavior of the indicator and different information it provides, interpretation it allows. The simplest way would be direct multiplication, i.e., *Area * BII*. Such indicator would signify that occupation of land with higher biodiversity intactness is a larger problem than occupation of land with low biodiversity intactness (the prior would result in higher values). On a first sight, this seems like the information we would want. Such approach would make sense for land use and land cover *change*. It would give the right information for deliberation on where to build a parking lot, to give a simple example. Nevertheless, this is not the question I want to answer here, and the data I use here do not directly capture land use change. Another option is multiplication by the factor of biodiversity intactness loss, i.e., *Area * (1-BII)*, which is the approach used, for example, by Hanafiah, Hendriks et al. (2012). As already discussed in Section 2.3, I believe this approach, and the interpretation it allows, is somewhat flawed. Furthermore, there is the fact that the footprint (land area) resulting from this characterization method is always smaller than the land area actually occupied, which to me seems a little problematic. (There is the same problem with the first approach, yet there it could be solved by adding one to the characterization factor, i.e., *Area * (1+BII)*.)

A third approach, the one I find the most adequate, is division by the characterization factor, i.e., *Area * (1/BII)*. Although it might not be directly apparent, such indicator has all the desired properties. It fulfills the first condition of environmental indicators that a larger number always means a more severe environmental impact. Another useful property of the indicator is that both land-sparing and land-sharing practices lead to improved results, i.e., the biodiversity footprint can be reduced by lower land occupation (land sparing) as well as improved biodiversity on the occupied land (land sharing). Also, it makes the land occupation appear more, rather than less severe – the resulting biodiversity footprint is always larger than the occupied land area. The condition that it should appear larger is not, strictly speaking, grounded in some physical fact – the real land area is still the same -, but I believe it adds to indicative power. This brings the question of what is the direct, physical interpretation of this indicator. Since BII is unitless, the results are expressed in a unit of area, square kilometer. Therefore, it does not directly express any biodiversity variable, like species richness, extinctions, abundance etc. Nevertheless, the same stands for almost every other methodology of biodiversity footprint, except for the one by Lenzen, Moran et al. (2012) where the unit is the number of threats (the limitations of this method were already discussed in Section 2.3). The only physical interpretation is the additional land needed to substitute the lost biodiversity by Hanafiah, Hendriks et al. (2012), which was already discussed. I see the indicator I present here as more of an extended land footprint than an outright indicator of biodiversity loss. Therefore, I believe an adequate interpretation of the indicator would be that it presents biodiversity-intactness weighted land occupation, or a biodiversity-intactness weighted area of land embodied in consumption.

4 Results

This section presents the results of the analysis of the biodiversity footprint coupled with the trade of products to the Czech Republic. First, the results of the total biodiversity footprint and the footprint divided by land use category for the last available year 2015 are presented, as well as the global country-distribution of the footprints. Then it is shown how the biodiversity footprint evolved between the years 1995-2015. Considering how significantly the footprint changed through this period, the results are further analyzed in the form of quartile averages. Thus, the regional and country distribution of the footprint, and its evolution in the time-period are presented for the total richness-based biodiversity footprint. Similarly, the respective contribution of the imported products and product categories and its evolution is analyzed. Finally, the trade balance (i.e., import minus export) is analyzed. Although I present a lot of numbers in the following sections, considering the uncertainty coupled with the results, those should be understood as orders of magnitude, rather than precise values.

4.1 Biodiversity footprint of import in 2015

Most footprint studies focus on analyzing the impacts in a single year. I will proceed similarly in this section, presenting the results for the latest available year 2015. Table 1 presents the aggregate results for import to the Czech Republic. In this year, the total "uncharacterized" land occupation, the total land area necessary to produce all the imported products as derived from the MRIO database, was 16,666 km^2. This is, for comparison, a little less than the area of Kuwait (17,818 km^2) and more than the area of Montenegro (14,452 km^2), and it is about 21.5% of the land area of the Czech Republic (Wikipedia 2022). The total richness-based biodiversity footprint, i.e., the occupied land area weighted by the richness-based Biodiversity Intactness Index, was 25,416 km^2. The abundance-based biodiversity footprint, slightly smaller than the richness-based, was quantified at 24,004 km^2. Figure 1, a) presents how the four land-use categories contribute to the total footprint. The largest part of the footprint is coupled with forest use. Nevertheless, the contribution differs between the three footprints as a result of the different biodiversity intactness factors connected with the land uses. As croplands are typically coupled with lower biodiversity than are forests, cropland use contributes to the total biodiversity footprint significantly more than to the land footprint. Compared to the forest and cropland use, which contribute around 50% and 45% respectively, pasture and other land contribute only marginally, less than 5% combined. The way in which the total footprint is divided between the land-use categories is almost identical for the richness-based and abundance-based footprint. Yet the characterization factors differ. The distribution of the difference between the richness-based BII and abundance-based BII factors (abundance *minus* richness) is illustrated in Figure 1, b). The difference values appear normally distributed with the mean of 0.051 (i.e., the abundance-based BII factors are on average larger by 0.051) and standard deviation of 0.051. There are some major outliers though. For example, the median abundance-based BII of forests in Finland is larger than the richness-based by 0.29. On the other side, the richness-based BII of forests in Norway is larger by 0.17. I have no robust explanation for such a large difference in character for the two geographically close countries. I only dare speculate that it has more to do with the data and the modelling procedure than the actual difference between Norwegian and Finnish forests. While I believe such outliers have little effect on the findings and conclusions of this analysis, it is a reminder that all the values should be treated with caution, since all the data used here are only the best available estimates based on incomplete evidence, rather than hard measurements.

Table 1 The 2015 results of land occupation, richness-based biodiversity footprint, and abundance-based biodiversity footprint coupled with imports of products to the Czech Republic. All the values are expressed in the unit of area (km^2).

2015	**Land**	**Richness**	**Abundance**
Cropland	6268 km^2	11476 km^2	10744 km^2
Forest	9736 km^2	12753 km^2	12134 km^2
Pasture	159 km^2	289 km^2	270 km^2
Other	504 km^2	897 km^2	856 km^2
Total	**16666 km^2**	**25416 km^2**	**24004 km^2**

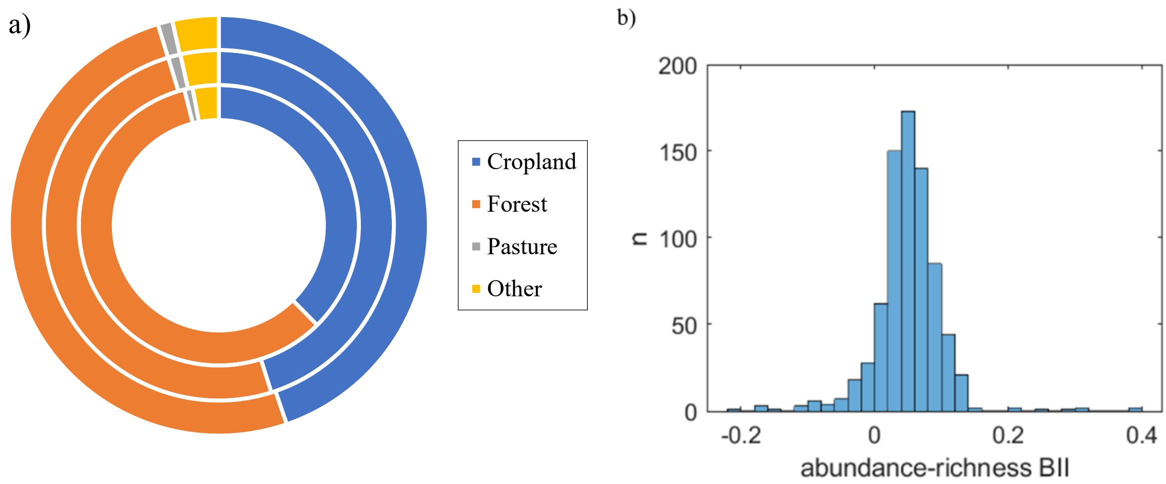

Figure 1 a) The contribution of the four land-use categories to the total footprint of imported products to the Czech Republic in 2015. The inner-most line represents "uncharacterized" land occupation, the middle represents the richness-based biodiversity footprint, and the outer represents the abundance-based biodiversity footprint. b) The distribution of the difference between richness-based and abundance-based BII characterization factors (abundance BII minus richness BII).

The maps in the Figures 2-4 illustrate the country distribution of the total biodiversity footprint in the year 2015. Figure 2 shows the distribution of the richness-based biodiversity footprint, and Figure 3 shows the abundance-based footprint. While the two footprints differ slightly, as it was discussed in the previous paragraph, the difference would not greatly affect the interpretation. Since it would be unpractical to present here twice all the almost identical maps and graphs, only the results for the richness-based biodiversity footprint will be analyzed further on. The maps show that the main bulk of imported biodiversity footprint is concentrated in the surrounding countries of central Europe. Indeed, the largest biodiversity footprint in 2015 was imported from the neighboring Poland (6724 km^2), followed by Slovakia (4706 km^2) and

Germany (1697 km^2). Other significant European countries in 2015 were Spain (1017 km^2), Hungary (839 km^2), and Croatia (663 km^2). The largest biodiversity footprint in non-European countries was imported from the African countries Cameroon (590 km^2) and Côte d'Ivoire (406 km^2), and from Indonesia 466 (km^2). The most important country of the Americas is Brazil (338 km^2). The significance of the European countries is further corroborated by the map in Figure 4, which displays the values of the richness-based biodiversity footprint normalized by the area of the country of origin (i.e., biodiversity footprint divided by the land area of the country). Here, the most affected country is Slovakia, followed by many other European countries – Poland, Croatia, Hungary, the Netherlands. While the "position" of Cameroon, Côte d'Ivoire, or Indonesia remains unchanged, the impact in Brazil seems insignificant.

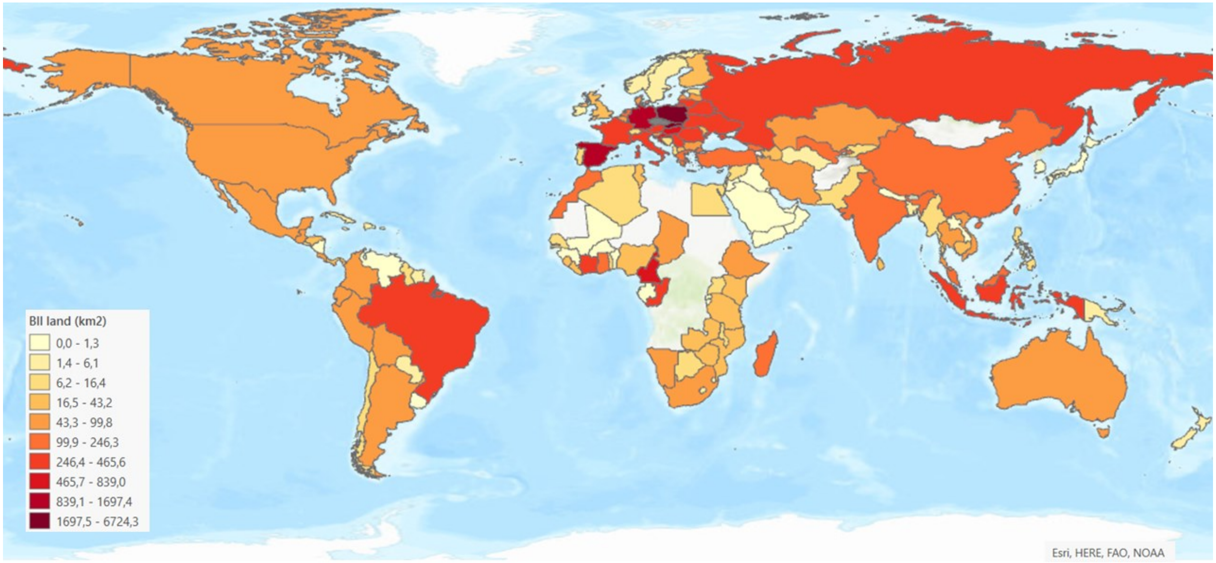

Figure 2 The map of country distribution of the total richness-based biodiversity footprint of products imported to the Czech Republic in the year 2015.

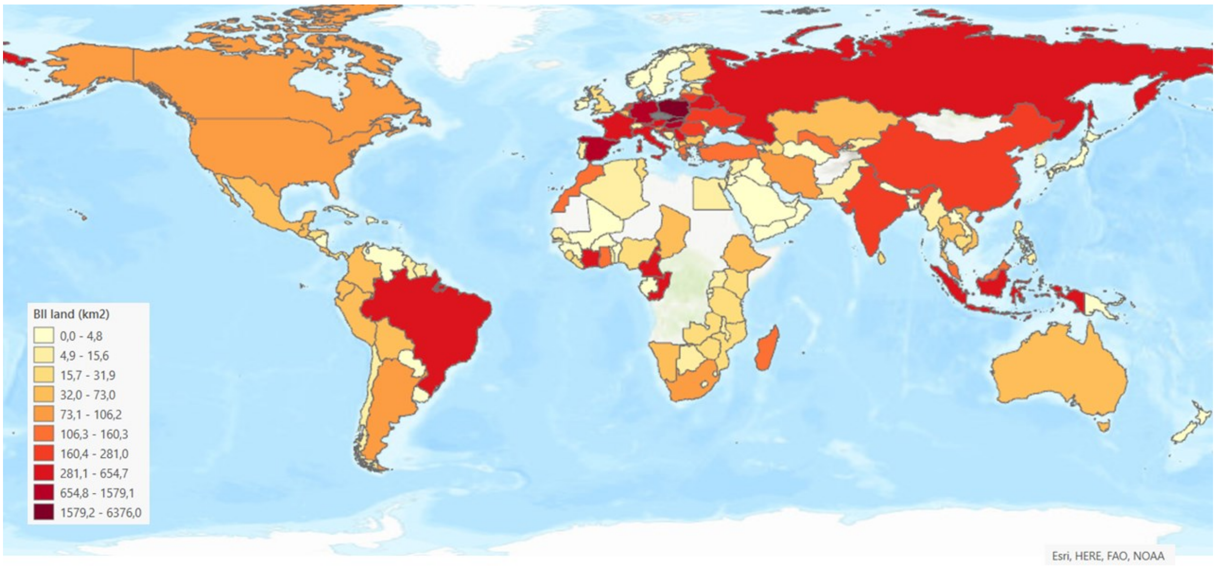

Figure 3 The map of country distribution of the total abundance-based biodiversity footprint of products imported to the Czech Republic in the year 2015.

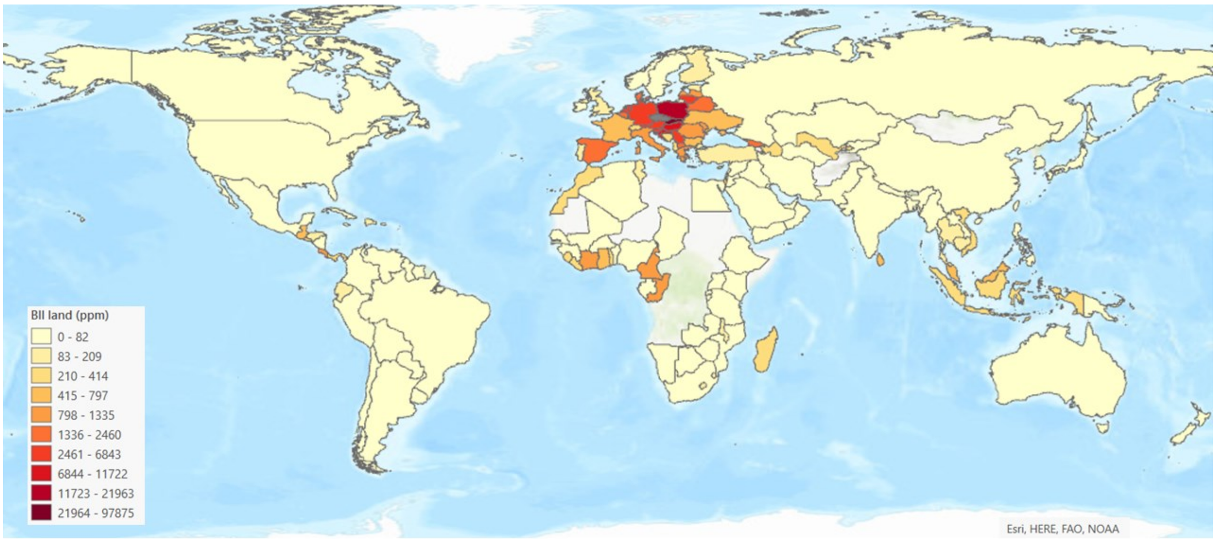

Figure 4 The map of country distribution of the total richness-based biodiversity footprint of products imported to the Czech Republic in the year 2015 normalized by the land area of each country. Results expressed in parts per million.

Figures 5-8 present the country distribution of the richness-based biodiversity footprint for the four land use categories. The map in Figure 5 shows the distribution of biodiversity footprint of the imported cropland. While the countries figuring in the "top positions" are the same as in the total footprint, their order is a little different. The largest cropland biodiversity footprint is imported from Slovakia (1907 km^2), followed by Poland (1154 km^2), Spain (999 km^2), and Hungary (718 km^2). Cameroon (584 km^2), Indonesia (441 km^2), and Côte d'Ivoire (404 km^2) figure here just behind Germany (604 km^2). By far the largest biodiversity footprint of forest use is imported from Poland (5173 km^2), as presented in Figure 6. The biodiversity footprint of forestry imported from Slovakia is half of that (2551 km^2). The next to most affected countries are again Germany (1000 km^2) and Croatia (596 km^2). New countries in this ranking are Belarus and Serbia (both 343 km^2). The non-European countries coupled with the highest biodiversity footprint of forestry are the Congo (330 km^2) and Brazil (151 km^2). The distribution of the biodiversity footprint of imported pastures, presented in Figure 7, looks much different, although this category contributes little to the overall footprint. The most affected country is Uzbekistan (73 km^2), followed by Namibia (44 km^2). With the exception of Slovakia in the third place (43 km^2), it is mostly countries like Zambia (27 km^2) or South Africa (21 km^2) that stand high in the order. I will discuss the validity of such results later (Section 5.1). In the other-land category (Figure 8), Poland (395 km^2), Slovakia (206 km^2), and Germany (89 km^2) stand again at the top of the ranking. In the fourth position now stand the Netherlands (41 km^2), followed by Brazil (23 km^2) and Lithuania (19 km^2).

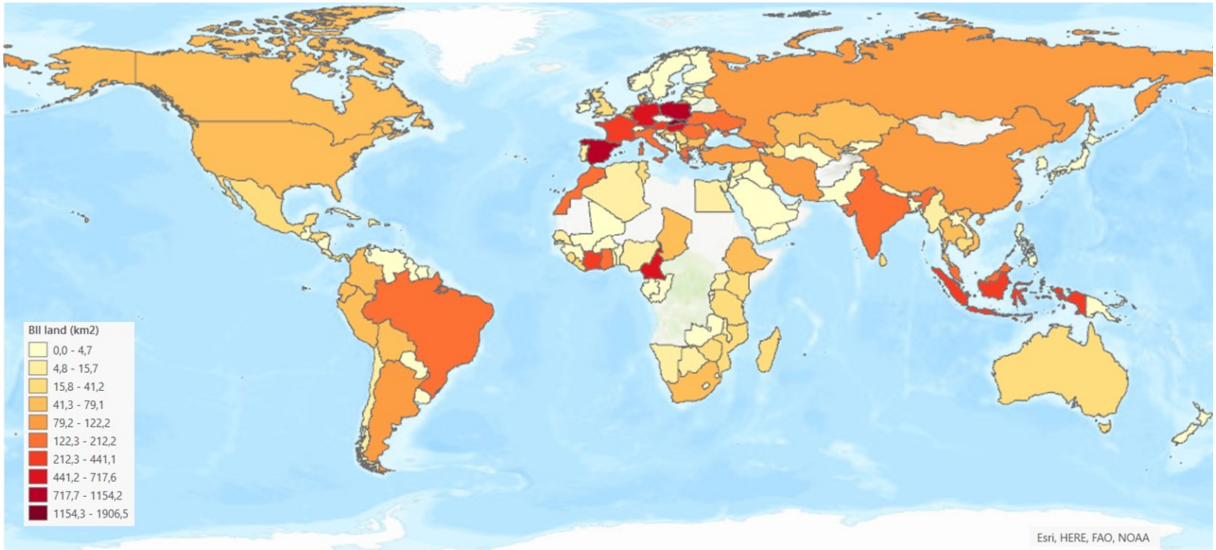

Figure 5 The map of country distribution of the richness-based biodiversity footprint of cropland use coupled with products imported to the Czech Republic in the year 2015.

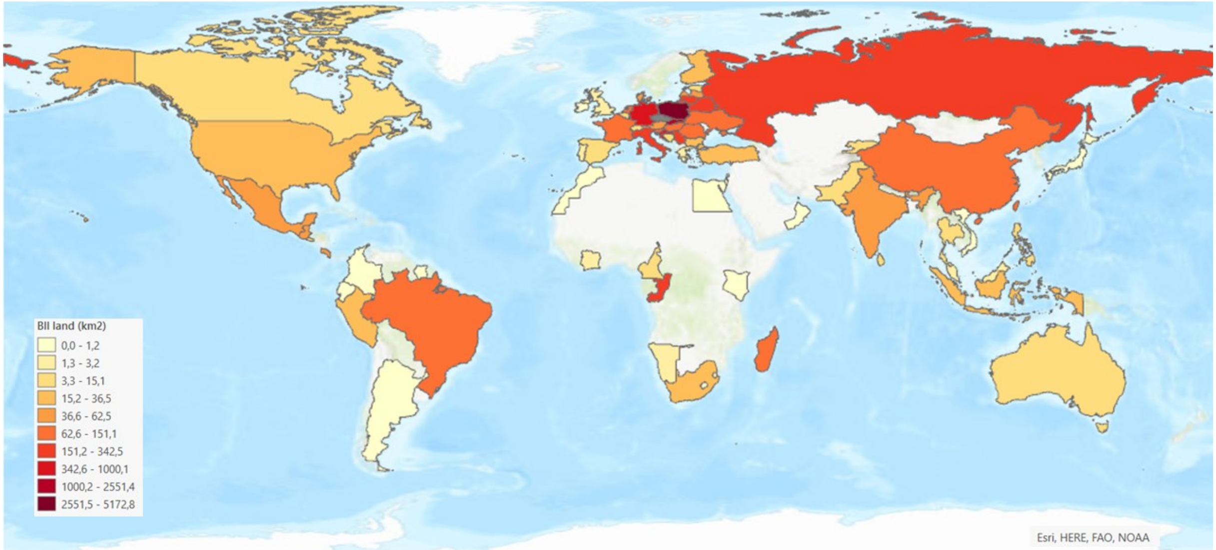

Figure 6 The map of country distribution of the richness-based biodiversity footprint of forest use coupled with products imported to the Czech Republic in the year 2015.

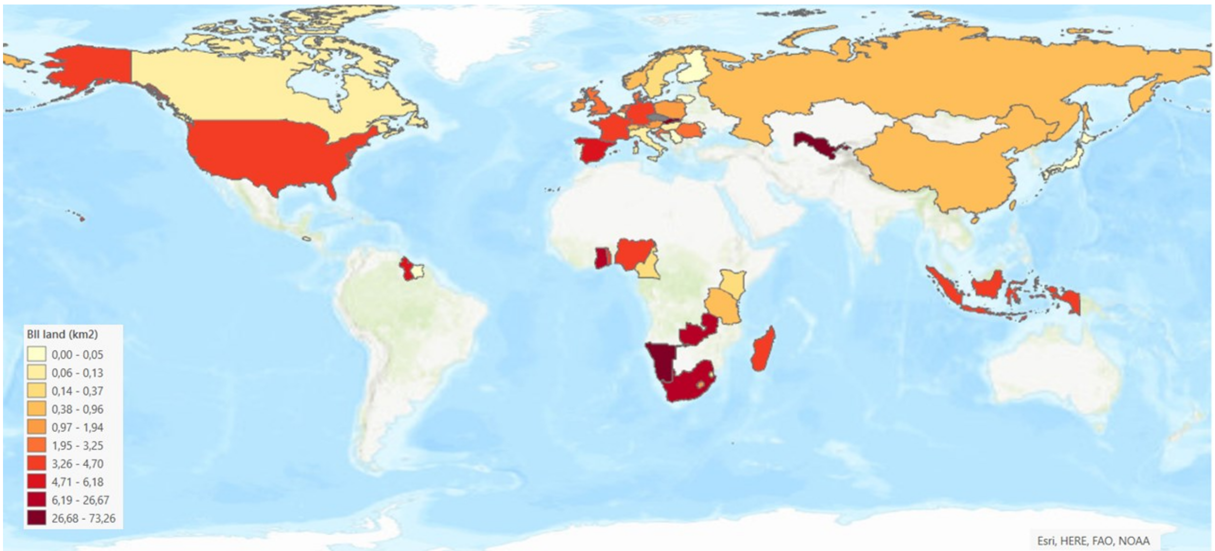

Figure 7 The map of country distribution of the richness-based biodiversity footprint of pasture coupled with products imported to the Czech Republic in the year 2015.

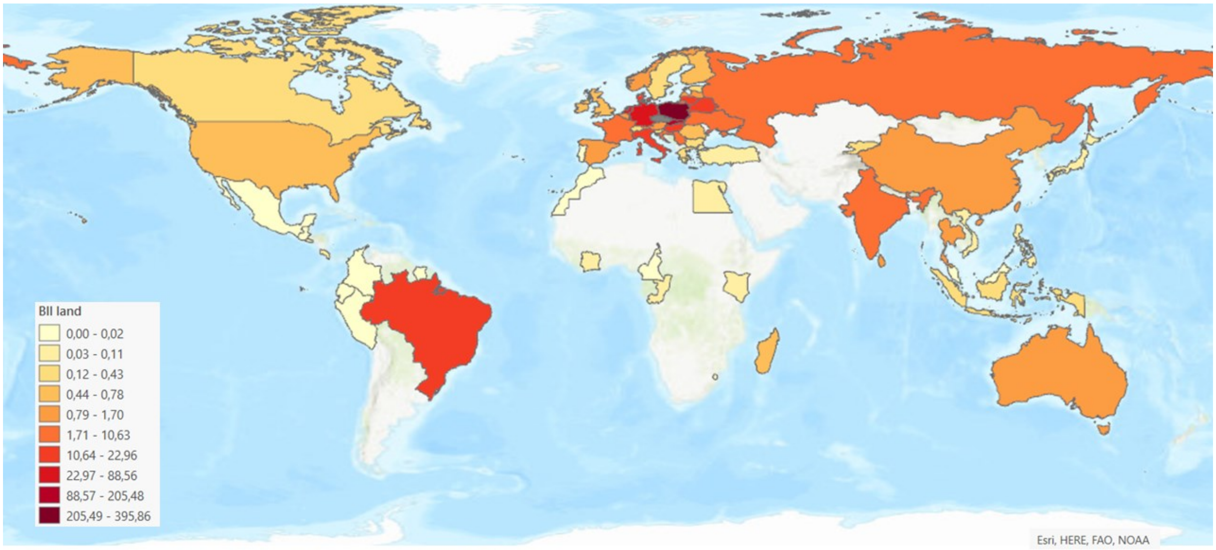

Figure 8 The map of country distribution of the richness-based biodiversity footprint of other-land use coupled with products imported to the Czech Republic in the year 2015.

4.2 Temporal evolution

Due to the complexity of constructing a global Input-Output table, it did not use to be practicable to evaluate long time series and most analyses were limited to single years. Yet the third version of EXIOBASE already allows multi-year analyzes. And temporality proves to be a crucial element. Figure 9 shows the evolution of the richness-based biodiversity footprint coupled with products imported to the Czech Republic between the years 1995 and 2015. In the total footprint, there can be identified four distinct segments. The period between 1995 and 2000 is characterized by slow growth of import, albeit with some oscillation. The period between the years 2000 and 2005 is characterized by a rapid growth of import, leading to doubling of the biodiversity footprint, from around 10,000 km^2 in the year 2000 to 20,000 km^2 in 2005. The growth in the period 2006-2010 slowed down, from the 20,000 km^2 in 2005 to around 26,500 km^2. The last quartile shows overall stagnation. Nevertheless, a slight drop after 2010, in the aftermath of the financial crisis of 2008, is apparent. There are signs of "recovery" and renewed growth in the year 2015, but the data end there. This pattern emerges mainly from the evolution in the categories of cropland use and forest use, although the trends for the two diverge. In the first quartile, the import of cropland footprint stays nigh on unchanged between 3000 km^2 and 4000 km^2. The trend for forest-use is more turbulent. Through large oscillations, the import of forest footprint climbs from around 1100 km^2 in 1995 to 5600 km^2 in the year 2000. It is also noticeable that in the first years, cropland was the dominant land-use, but forest use catches-up by the turn of the millennium. In the period between 2000 and 2005 the import of forest footprint oscillates with a significant upward trend. On the other hand, the cropland footprint, after a major leap in the year 2001, remains stagnant. Therefore, in 2008 the import of forest footprint overtakes cropland and remains so until the end of the analyzed period. Around the year 2010, after the financial crisis, the trend of the two land uses decouples. While forest footprint grew in 2009, cropland footprint dropped, and vice versa since the year 2010. Since the year 2014, both cropland and forest biodiversity footprints seem to grow again.

The trends in the import of pasture and other-land footprints are different again. The other-land footprint continuously grows through the entire time-period, from 18 km^2 in 1995 to 897 km^2 in 2015. In contrast, the biodiversity footprint of embedded-pasture import grows in the first half from 250 km^2 in 1995, peaks in the year 2003 at 2490 km^2, and then recedes back to 290 km^2 in 2015. From Figure 9 it is apparent that the total richness-based biodiversity footprint of products imported to the Czech Republic was nearly six times larger in the year 2015 than in 1995. It is also clear that which year is selected for analysis can significantly affect some conclusions. For example, in the year 1995, cropland makes 67% of the total footprint, while forest makes only 27% and pasture makes 6%. In 2003, cropland makes 46% of the total footprint, forest makes 37%, and pasture 16%. In comparison, in 2009, forest-use makes 66% of the total footprint, cropland 29%, and pasture only 3%. Based on this, the results further on will be presented in the form of quartile averages – 1995-2000, 2001-2005, 2006-2010, and 2011-2015. These quartiles, next to being quite an obvious way to separate the time-period, also encompass the four distinct segments of the total biodiversity footprint trend described above.

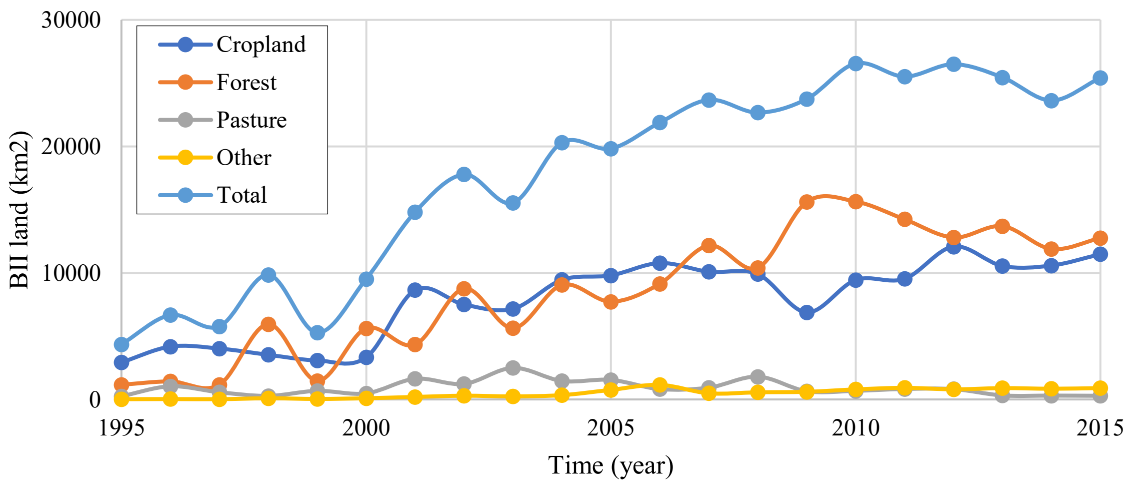

Figure 9 The temporal evolution (1995-2015) of the richness-based biodiversity footprint coupled with products imported to the Czech Republic.

4.3 Geographical distribution of the impacts

In this section I present how the country-distribution of the imported richness-based biodiversity footprint evolved in time. It was described in the previous section how the footprint fluctuated from year to year. The variety of the countries of origin is even greater, especially for African countries. It is common that some countries are a major source of the imported biodiversity footprint in one year yet remain unaffected in other years. An extreme case is the Democratic Republic of the Congo, which is a source of over 9000 km^2 of biodiversity footprint in 2009, over 6200 km^2 in 2007, but nothing is imported in 2014 or 2015. Most of this footprint is coupled with products of forestry, and the validity of such results will be discussed later (Section 5.1). Similar pattern with several extreme values can be observed, among others, for South Africa, Kazakhstan, the Congo, or Belarus. This extreme variance is partly smoothed by the quartile-averaging. Therefore, the quartile averages represent the overall patterns much more robustly than single-year results.

Figures 10-13 show the country distribution of the quartile-average richness-based biodiversity footprint. Common to all quartiles, and same as in the 2015 results presented in Section 4.1, is the pattern that the neighboring countries Poland, Slovakia, and Germany hold a prominent position. Nevertheless, they are not so dominant as in the 2015 results. In the first quartile, 1995-2000, the largest average biodiversity footprint is imported from Germany (624 km^2) and Slovakia (509 km^2). Those are followed by Uzbekistan (399 km^2), as a major source of plant-based fibers (cotton), and meat. In the fourth position is China (385 km^2) with a vast portfolio of products, mainly oil seeds and products of forestry. Fifth is Côte d'Ivoire (338 km^2) with product category *Vegetables, fruit, nuts* and also unspecified crops (*Crops nec*), cocoa is likely the main product. Sixth is the Democratic Republic of the Congo (337 km^2) where the main biodiversity footprint is coupled with meat animals. Among the most affected countries are also Indonesia (251 km^2), with unspecified crops and products of forestry, Mexico (218 km^2) as a source of products of forestry, or Argentina (205 km^2) as a source of oil seeds and products of forestry. Other notable flows are forest products from Romania, plant-based fibers from Tajikistan and Turkmenistan, or crops from Nigeria.

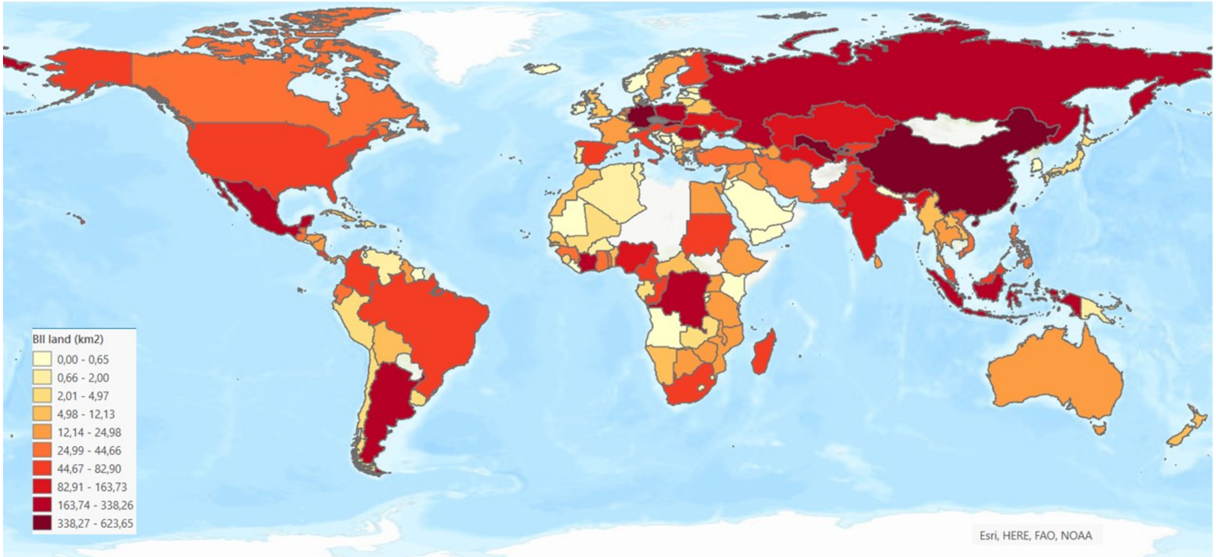

Figure 10 The map of country distribution of the average total richness-based biodiversity footprint coupled with products imported to the Czech Republic in the years 1995-2000.

In the second quartile, presented in Figure 11, Slovakia (1673 km^2) and Germany (1176 km^2) are joined by Poland (1256 km^2) in the top positions. These are followed by Kazakhstan (789 km^2) as a source of meat animals and plant-based fibers, the Congo (615 km^2) as a source of meat animals and products of forestry, Uzbekistan (538 km^2) as a source of plant-based fibers, and Indonesia (507 km^2) as a source of unspecified crops and products of forestry. The fact that some countries figure lower in the order than before does not necessarily mean the footprint decreased, but rather that it grew less rapidly than some others. An example is China that fell from fourth to tenth position, yet the footprint grew from 385 km^2 to 442 km^2. On the other hand, the biodiversity footprint in Mexico did in fact drop from 218 km^2 to 186 km^2. Some other notable flows in this period are vegetables, fruits, and nuts from Spain, in some years there were large flows of products of forestry from Madagascar or Belize, crops from Nigeria or Malaysia, or oils seeds from China. There was also a notable import of oil seeds and products of forestry from the United States.

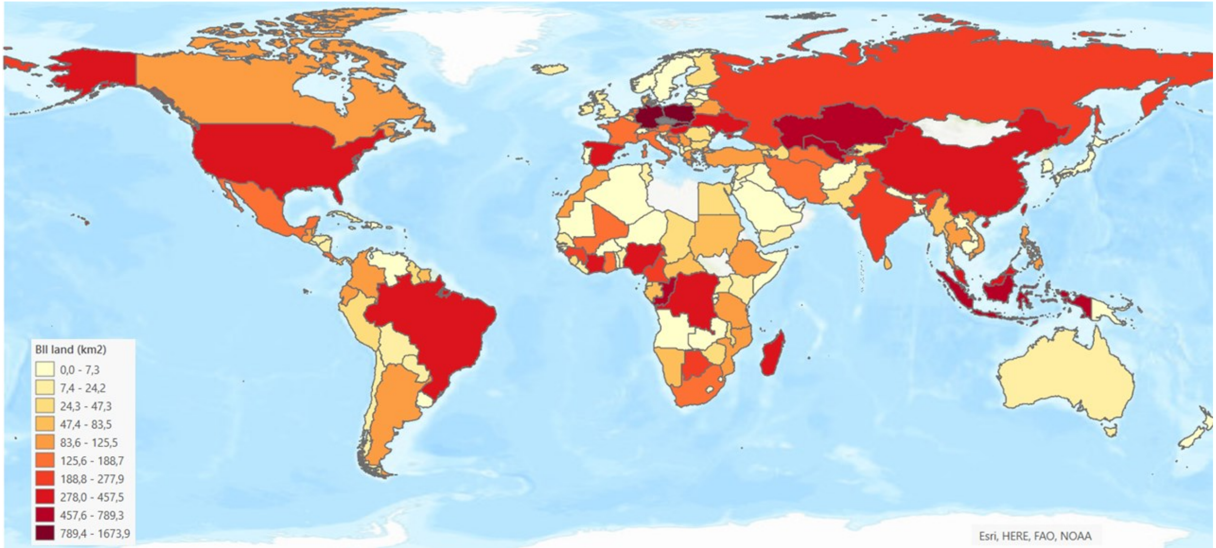

Figure 11 The map of country distribution of the average total richness-based biodiversity footprint coupled with products imported to the Czech Republic in the years 2001-2005.

The results for the third quartile, presented in Figure 12, are dominated by the biodiversity footprint coupled with products of forestry from the Democratic Republic of the Congo (5331 km^2), followed by the usual trio of Slovakia (2420 km^2), Poland (1414 km^2), and Germany (1251 km^2). The new country in the top positions is Belarus (733 km^2) as a result of one extreme flow of products of forestry in 2010 (3145 km^2). The other countries remain largely similar as in the previous quartile, with the notable exception of China which dropped further down the list. Côte d'Ivoire (553 km^2) remains an important source of vegetables, fruit, nuts and unspecified crops (cocoa). Uzbekistan (543 km2) remains an important source of plant-based fibers (cotton), but in some years the flows of meat animals and products of forestry are larger. A large biodiversity footprint is also coupled with the import of vegetables, fruit, and nuts from Cameroon. Again, crops from Indonesia and vegetables, fruit, and nuts from Spain are major flows. There are also consistent flows of vegetables, fruits, nuts, and unspecified crops from Brazil. Compared to the previous quartile, the import of biodiversity footprint coupled with products of forestry from Russia grew significantly.

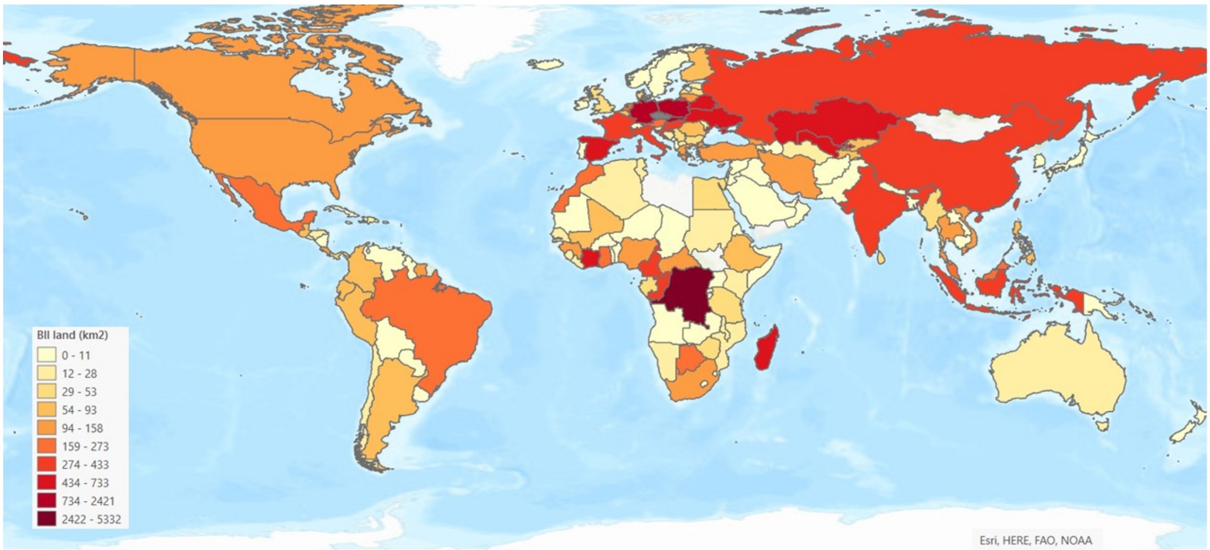

Figure 12 The map of country distribution of the average total richness-based biodiversity footprint coupled with products imported to the Czech Republic in the years 2006-2010.

The main change in the last quartile (Figure 13) is that the flows of products of forestry from the Democratic Republic of Congo dissipated. Therefore, Poland (5406 km^2), Slovakia (3914 km^2), and Germany (1802 km^2) returned back to the very top. Out of nowhere emerged the Central African Republic (909 km^2) with major imports of products of forestry. In this quartile further grew the biodiversity footprint coupled with imports from Spain (906 km^2), mainly vegetables, fruit, nuts, and oil seeds. The two Congos remain high on the list with the imports of products of forestry. Hungary (681 km^2), with imports of oil seeds and cereal grains, climbed higher than in the previous periods. The biodiversity footprints in Côte d'Ivoire, Cameroon, or Indonesia remain high in the order.

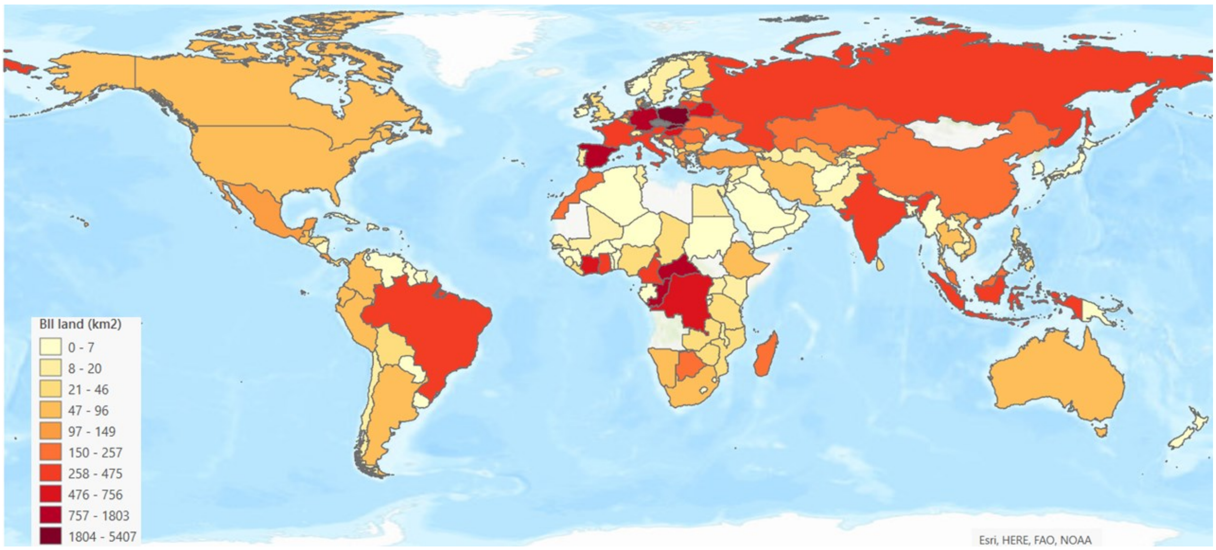

Figure 13 The map of country distribution of the average total richness-based biodiversity footprint coupled with products imported to the Czech Republic in the years 2011-2015.

4.4 Product-category distribution

In this section I present how the various products (product categories) contribute to the richness-based biodiversity footprint of import to the Czech Republic. The results for all the land-use categories combined are presented in the column charts in Figure 14. The figure is divided into two parts for the sake of visibility, since the magnitude of the respective biodiversity footprints largely differs. The results are presented separately for each quartile. Thus, it is also shown how the import of the product-categories evolved through the period of analysis (1995-2015). From the chart it is clear that the largest biodiversity footprint is coupled with the import of *Products of forestry, logging and related services*, and the dominance gets even more pronounced with time. The origin of this footprint evolves in time as well. In the first quartile, the products of forestry are imported mainly from European countries like Germany, Slovakia, or Romania, but there is a notable import also from China or Mexico. The major growth of import in the following years was, next to the neighboring countries of Europe, provided by countries like Belarus and Russia, but also by more exotic countries, like the Congos or Madagascar. It should be stressed that those are products of forestry embedded in the imported products, i.e., the wood needed to produce those products, including fuelwood. Therefore, it is likely that a large part of the products of forestry never physically made it to the Czech Republic as timber, but were used to produce imported goods, for example, from China. I believe this point is more relevant for the products of forestry than for other product categories.

The second most-contributing product category, in the first quartile, is the category *Plant-based fibers*, imported mainly from the central Asian countries of Uzbekistan, Turkmenistan, Tajikistan, and Kazakhstan. Nevertheless, after a minor growth in the second quartile, this footprint drops rapidly. One possible explanation is that plant-based fibers were increasingly substituted by synthetic fibers. On the other hand, the global production and consumption of cotton still slightly grew in this period (OECD and FAO 2021). Since it is exactly the footprint in the cotton-producing Central-Asian countries that dissipated in the third and fourth quartiles, I would attribute this to some other factors, mainly data deficiency. The third category in the first quartile is the *Crops nec*. It is a group of crops that are not further specified; hence, this category is dominated by the so-called developing countries, i.e., countries where the trade statistics are less detailed. The biodiversity footprint in this product-category does not grow much, even though the overall cropland footprint grew (Figure 9). Therefore, I believe the relative lack of growth could be explained by an increased detail in production statistics. Indeed, the biodiversity footprints of *Oil seeds*, *Cereal grains nec*, *Wheat*, or *Sugar cane, sugar beet* product-categories grew rapidly.

Since the second quartile, *Vegetables, fruits, nuts* has been the second most-contributing product category. In the first quartile, the biodiversity footprint was located mostly in Spain, Côte d'Ivoire, and Columbia. In the second quartile, Columbia was substituted by Poland at the position of a major importer, but there were many other countries coupled with import of this product-category. Next to the already mentioned countries, Cameroon and Ghana became major importers in the third and fourth quartiles. The import of biodiversity footprint coupled with production of *Oil seeds* grew nearly six-fold in the assessed period. In the first quartile, oil seeds came mainly from China and Argentina, both major soybean-producing countries (OECD and FAO 2021). In the following quartiles, the import of (likely) soybean from China, USA, or India was accompanied and overtaken by the biodiversity footprint of oilseed imported from Slovakia, Germany, Spain, or Poland. This growth converges with the massive global growth of production and consumption of biofuels (OECD and FAO 2021). The biodiversity footprint coupled with the import of cereal grains and wheat grew even more rapidly. Cereal grains and wheat came consistently mainly from Slovakia, Hungary, Germany, Poland, and increasingly from France. A large growth is also apparent in the category *Sugar cane, sugar beet*, albeit the contribution to the total footprint is small. This sugar comes mainly from Slovakia, Austria, Poland, and Germany. A surprisingly small biodiversity footprint appears to be coupled with rice – very rough estimates based on data from FAOSTAT and from the Czech Statistical Office suggest it should be about 100-times more. I believe this to be a result of the (lack of) detail in the trade statistics, i.e., most of the imported rice is included in the category *Crops nec*.

A surprisingly small biodiversity footprint is also coupled with import of animal products, i.e., the product categories *Meat animals nec*, *Cattle*, *Poultry*, and *Pigs*. The biodiversity footprint of animal products was smaller than that of crops and vegetables in all quartiles. Based on the data, the largest biodiversity footprint coupled with meat animals was imported from countries like Kazakhstan, the Congos, Ghana, Botswana, or Namibia. Nevertheless, there are many countries with small flows, rather than a few countries with large flows. According to the Czech Statistical Office (e.g., *www.czso.cz/csu/czso/ari/agriculture-4-quarter-and-year-2015*) meat is imported mostly from Poland, Slovakia, Germany, and other European countries. It is possible that the crops used to feed the meat animals originate in those distant countries like Kazakhstan, but the fact that the associated land use is pasture contradicts this. Therefore, while the total amount of imported meat animals in the MRIO database is probably correct (as it is based on national trade statistics), the assigned countries of origin (which affects the land and biodiversity footprints) are likely mislocated as a result of the database construction procedure. The biodiversity footprint linked to the specified animal-categories (*Cattle*, *Poultry*, *Pigs*) is, indeed, imported mainly from Germany, Slovakia, Poland, or Denmark.

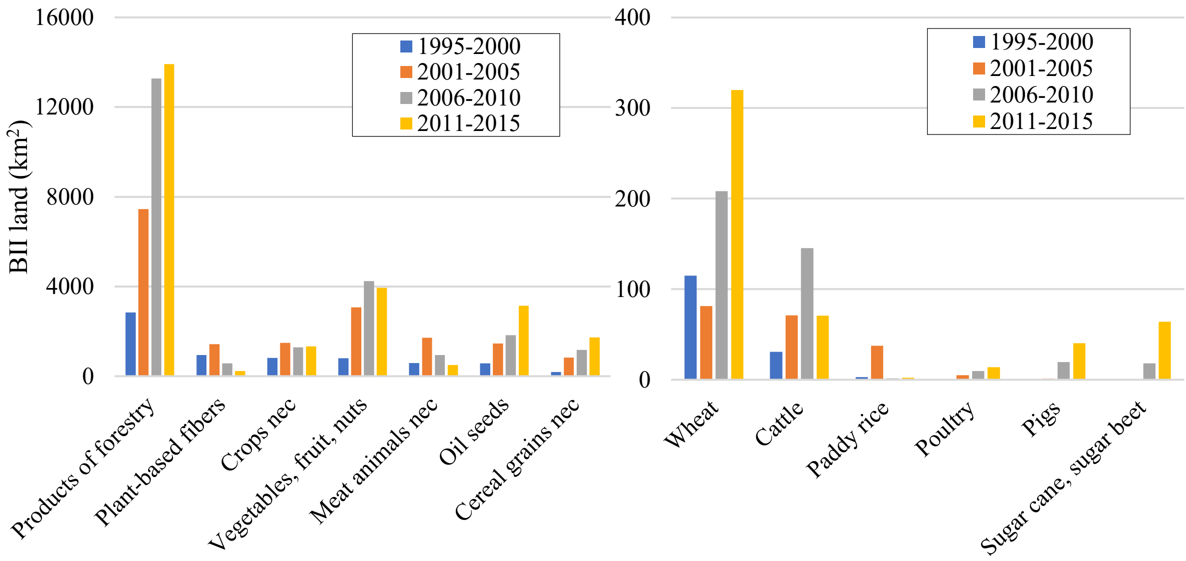

Figure 14 Contribution of the various product categories to the richness-based biodiversity footprint embedded in products imported to the Czech Republic, all the land-use categories combined. The results are divided into two separate column charts for the sake of visibility.

4.5 Trade balance

In the previous sections I analyzed the biodiversity footprint embedded in products imported to the Czech Republic. Those results indicate a rapid growth of impacts in the assessed period (1995-2015), a little too rapid to realistically correspond to a growing consumption. However, it is critical to remember that those results do not represent all the impacts coupled with domestic consumption, but only the internationally traded goods. The analyzed period represents a time following a fundamental change in the economic system of the Czech Republic after the fall of the Communist regime, characterized by implementation of (neo)liberal economic policies in the 1990s. The market liberalization was occurring in most countries of the world though, making the global economy increasingly interconnected. Furthermore, in the early 2000s, the Czech Republic joined the EU single market, which facilitated trans-boundary flow of goods even further. I believe the liberation of boundaries in international trade is the main reason behind the ballooning biodiversity footprint displayed in the previous section, and the trade-balance results corroborate this hypothesis. A typical feature of the global economy, sometimes called the resource curse, is that some (often poor) countries export raw resources, and other (rich) countries export high-tech, high added-value goods made from those resources. Land footprint and biodiversity footprint indicators, as employed in this study, depict land and biodiversity as one of the globally traded limited resources. Indeed, most footprint studies demonstrate the same pattern: "developing" countries export limited environmental resources (footprints) to rich countries (Weinzettel, Hertwich et al. 2013, Wiedmann and Lenzen 2018). This section aims to show where does the Czech Republic stand in this dichotomy. To understand the findings properly, it is important to remember that the biodiversity footprint in this study represents relative, not absolute biodiversity, i.e., the intactness of all global ecosystems is valued equally.

The trade balance in this study is understood as the difference between the imported biodiversity footprint and the exported biodiversity footprint, i.e., import *minus* export. Hence, values over zero mean that the imported biodiversity footprint is larger than the exported, and vice versa. Figure 15 shows a very turbulent evolution of the balance of trade in the richness-based biodiversity footprint for the Czech Republic. In 1995, the Czech Republic imported more biodiversity than it exported, and it was so for all the land-use categories. In 1998 the scales started to tip, and in 1999 the Czech Republic exported larger biodiversity footprint than it exported. This was connected mainly with the increased exports of cropland. After a further dip in the year 2000, the balance returned to positive numbers. The positive balance for biodiversity footprint in 2004 here is contradictory to the negative land footprint balance in that year presented by Steen-Olsen, Weinzettel et al. (2012). In 2005 the balance turned again following an upsurge of export, which more than doubled. In the following years, the trade was nigh on symmetrical, until a further drop started in 2010. The export in all land-use categories exceeded the import for the first time in 2013. In the year 2015, the balance hit a record low at a deficit below -20,000 km^2. The biodiversity footprint coupled with cropland is the main driver of the overall balance, and so the trend for cropland is largely similar to the total. The export of the biodiversity footprint coupled with forest-use grew continuously throughout the entire period. Therefore, the tumultuous trend in the balance is mainly driven by the oscillating rate of import. The trade in the biodiversity footprint coupled with the other-land category was close to an equilibrium, albeit with a downward tendency. Regarding pasture, larger footprint was imported in the first half of the assessed period. Around the year 2005 the balance started to turn, though, and in the second half of the period, a larger footprint was exported from the Czech Republic than imported.

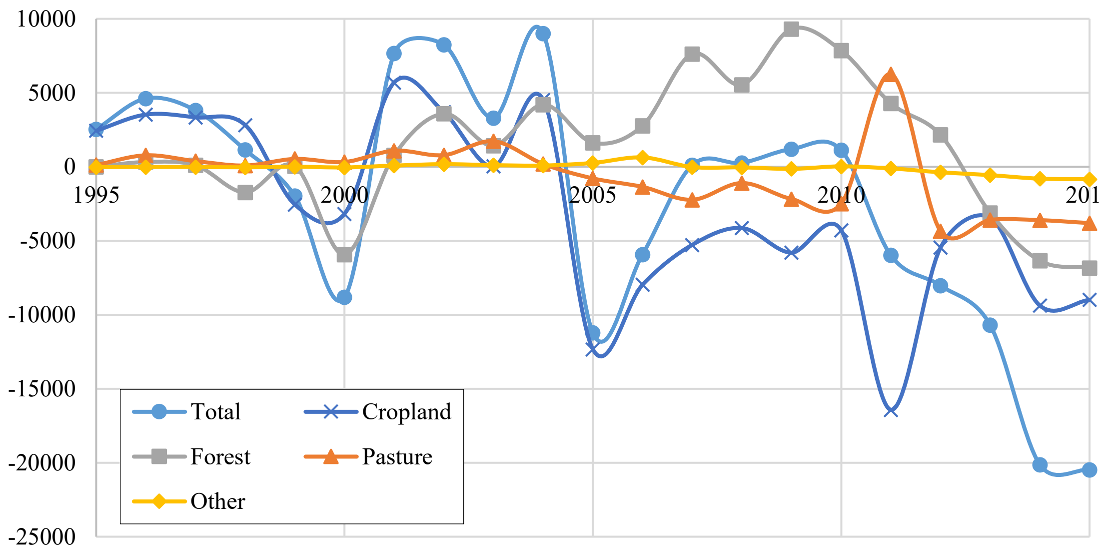

Figure 15 Time evolution of the richness-based biodiversity footprint trade balance for the Czech Republic between the years 1995-2015.

Figure 16 presents the trade balance for the various product categories in the period 1995-2015 divided to quartiles; and the relative status of the products is entirely different than for import (Figure 14). Significantly positive numbers (i.e., import larger than export) are apparent for *Vegetables, fruit, nuts* and for *Crops nec* categories. This is hardly surprising, production of many kinds of vegetables, fruit, and nuts (e.g., citruses, bananas), many kinds of crops (e.g., cocoa), and plant-based fibers (cotton) simply is not possible or profitable in the Czech Republic due to the climatic conditions. On the other hand, large flows of biodiversity footprint coupled with *Vegetables, fruits, nuts* category were exported as well, especially to Slovakia. The Czech Republic is a net exporter of the biodiversity footprint coupled with production of cereal grains. Indeed, the trade difference is by far the largest for wheat. The richness-based biodiversity footprint of wheat exported from the Czech Republic is more than eight times larger than the imported. The biodiversity footprint of cereal grains and wheat was exported mainly to Germany and Poland, but also to countries like Algeria, Morocco, or Jordan. The biodiversity footprint of *Oil seeds* starts with positive numbers in the first quartile, which I believe is driven mainly by the import of soybean discussed in the previous section. Nevertheless, this changes in the second quartile, and the export of oil seeds significantly exceeds import, especially in the third quartile. This is likely connected with the boom of the infamous rapeseed grown for production of biodiesel. Oil seeds were exported mainly to Germany, Russia, and Poland. The trade in biodiversity footprint coupled with *Sugar cane, sugar beet* is close to being balanced. It starts with slightly negative numbers and turns positive in the second half of the assessed period. As discussed in the previous section, most of the rice imports are likely concealed in the *Crops nec* category; thus, the trade balance for *Paddy rice* footprint appears close to equilibrium even though the Czech Republic exports no rice.

The trade is balanced also for *Pigs* and *Poultry* - the exported richness-based biodiversity footprint for those product categories slightly exceeds the imported footprint. While the imported footprint of *Meat animals nec* is significantly larger in the first two quartiles, in the last quartile a larger footprint is exported. In the last two quartiles, the largest flows of biodiversity footprint coupled with meat animals were exported to Turkey, Kazakhstan, or the USA. In the previous section, I wondered about the results that indicated that countries like Kazakhstan are a major source of meat for the Czech Republic. Nevertheless, even larger flows of biodiversity footprint of meat were apparently exported to many of those countries at the same time. The trade balance of cereal grains can also elucidate the seemingly small imports of meat animals from Germany or Poland. It is possible that a large part of the feed for the animals imported from those countries originated in the Czech Republic. The negative trade balance for cattle is less enigmatic, because the Czech Republic exports large amounts of milk (e.g., *czso.cz/csu/czso/ari/agriculture-4-quarter-and-year-2015).* While the biodiversity footprint embedded in *Products of forestry, logging, and related services* loomed over the others when only import was considered, when balanced against exports it does not appear so outstanding, with the notable exception of the third quartile. In the first quartile, the exported biodiversity footprint of forestry exceeded imports. In the second and third quartiles, the imports grew rapidly while exports stagnated. In contrast, the rate of import grew little in the last quartile, but exports soared and exceeded the imports again. Products of forestry were exported predominantly to Germany and Austria, but there were also major exports to China, especially in the last quartile.

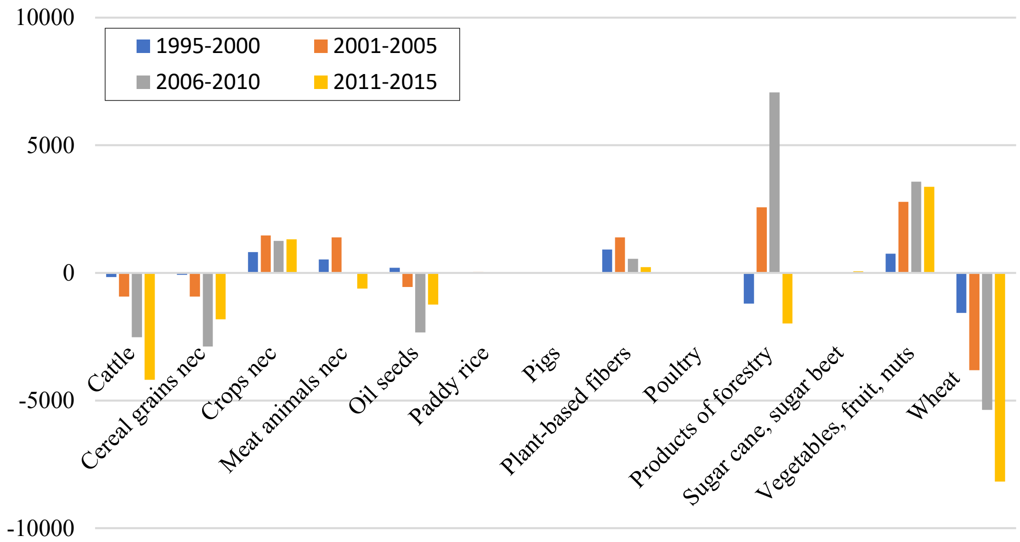

Figure 16 The balance in the Czech trade of the richness-based biodiversity footprint for the various product categories, in the period 1995-2015.

The overall results (Figure 15) indicate that the Czech Republic shifted from a net importer of biodiversity footprint in the first half of the analyzed period, to a net exporter of biodiversity footprint. Based on the pattern described at the beginning of this section, this would indicate that the Czech Republic used to be a "developed" country in the 1990s but became a "developing" country at latest by the year 2010. Clearly, this is quite a preposterous suggestion. The Human Development Index (HDI) of the Czech Republic – although limited and western-centric in its conception of "development" - did not only grow in the absolute terms, but the country climbed significantly in the ranking (UNDP 2020). One way to explain this is that although the Czech Republic is a relatively rich and "developed" country, it is still exploited by a few countries that are even richer and more "developed." Another possible hypothesis could be that the developed/developing dichotomy uncovered in previous biodiversity footprint studies (Wiedmann and Lenzen 2018) is largely driven by the fact that the "developing" countries are commonly situated in the tropics, where more species of organisms live. Furthermore, "development" (as commonly understood) is inherently coupled with destruction of natural ecosystems; thus more "developed" usually means there is less space for nature. Therefore, any activity has a disproportionally larger biodiversity footprint in the biodiversity-rich countries if the indicator is based on absolute metrics (e.g., the number of species). Contrarily, the biodiversity footprint, as measured here, is larger for activities occurring in places with lower relative intactness, hence the dichotomy might be abated. Unfortunately, to be able to evaluate what is the relative weight of those, and potentially other, factors is beyond the capability of this study, because it would be necessary to evaluate similar data for other countries.

The maps in Figures 17-20 present the balance in the bilateral trade with the richness-based biodiversity footprint for the Czech Republic and other world countries. In the first quartile (1995-2000), the Czech Republic was a net importer of the total biodiversity footprint, which is apparent on the map (Figure 17), as only a few countries are displayed in green. The "green" countries are predominantly situated in Europe; the largest average trade "deficit" occurred with Austria (-2548 km^2), Algeria (-914 km^2), Belarus (-498 km^2), and Germany (-269 km^2). The colors of the picture indicate some north/south, east/west divide, with the largest "surplus" occurring for Uzbekistan (399 km^2), Slovakia (347 km^2). Côte d'Ivoire (338 km^2), The Democratic Republic of Congo (337 km^2), Romania (259 km^2), Indonesia (242 km^2), and many other arguably less "developed" countries. On the other hand, there are also several richer countries that exported more than they imported, such as Finland (66 km^2), Spain (54 km^2), or the USA (49 km^2). The map for the second quartile (Figure 18) is noticeably "greener," although the total five-year average trade balance is still significantly positive (3399 km^2), and the number of countries that export more to the Czech Republic is larger as well (137 to 37). The largest trade deficit in this quartile occurred with Austria (-3252 km^2), Germany (-1444 km^2), Jordan (-1290 km^2), and Algeria (-1023 km^2). On the other side, the largest surplus occurred with Slovakia (1133 km^2), the Congo (615 km^2), Kazakhstan (548 km^2), Uzbekistan (538 km^2), Indonesia (489 km^2), and other more eastern or more southern countries. Again, there are the cases of the USA (259 km^2), Spain (115 km^2), or France (115 km^2) that disrupt the poor-to-rich pattern. The third quartile (Figure 19) appears largely similar to the second, only a little greener. The number of countries that export more is still markedly higher (126 to 42), although the overall five-year average balance is now in deficit (-620 km^2). The main notable difference is that the previous surplus in the trade with the USA now turned into a slight deficit (-18 km^2), but France (206 km^2) and Spain (138 km^2) still disrupt the pattern. The last quartile (Figure 20) is apparently even greener. While the ratio of the "red" and "green" countries remains almost the same, the overall average balance is now in a deficit of -12,986 km^2. An interesting case is China, which now changed from a major net exporter (in relation to the Czech Republic) to a net importer of biodiversity footprint (-714 km^2), and the same stands for Kazakhstan (-276 km^2).

While the overall trade balance for the richness-based biodiversity footprint contradicts the developing/developed dichotomy, a pattern does emerge when bilateral trade balance is analyzed. Most tropical "developing" countries export larger biodiversity footprint to the Czech Republic than they import, and the richer neighbors Germany and Austria import significantly more than they import. Yet there are also many cases that strongly contradict this pattern, especially Spain or France on the one side, and Jordan, Algeria, and later China on the other side. The number of countries that import more than they export is not that far from the position of the Czech Republic in the HDI ranking (UNDP 2020), but while the number of more "developed" countries decreased during the period (from 34 to 26), the number of countries the Czech Republic had a deficit with rose (from 35 in the first quartile to 43 in the last quartile). Furthermore, there is next to no correlation (r=-0,14) between the average trade balance (in the last quartile) and the per capita gross domestic product in 2015 (TheWorldBank 2022). In conclusion, there are some signs of the exploitative pattern emerging in the results of the trade for richness-based biodiversity footprint for the Czech Republic, but the pattern is weak and in no way universal.

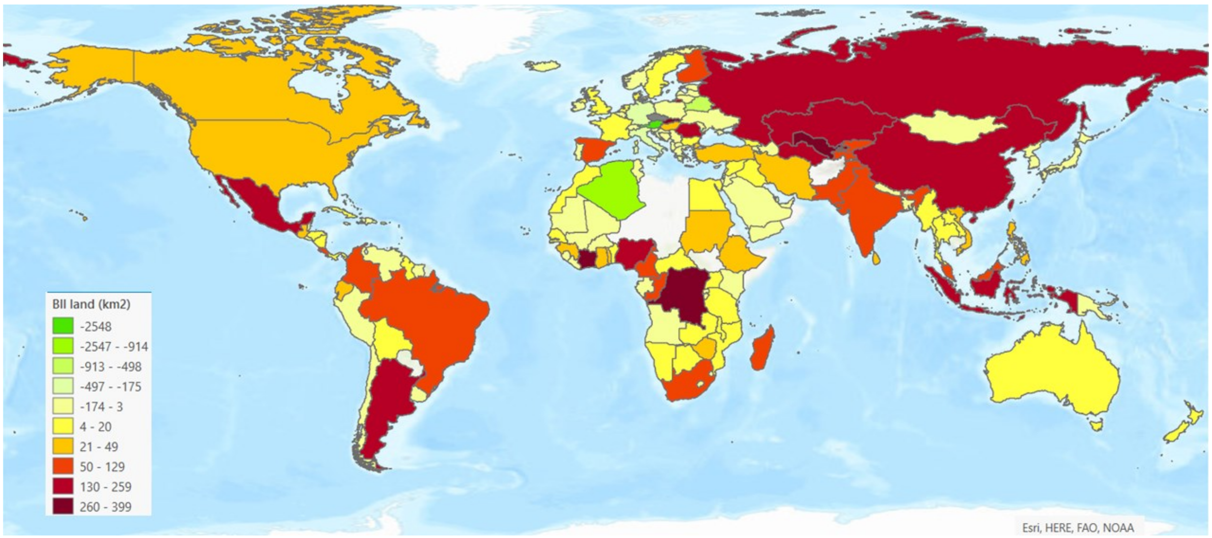

Figure 17 The map of country distribution of the trade balance in the total richness-based biodiversity footprint for the Czech Republic in the years 1995-2000.

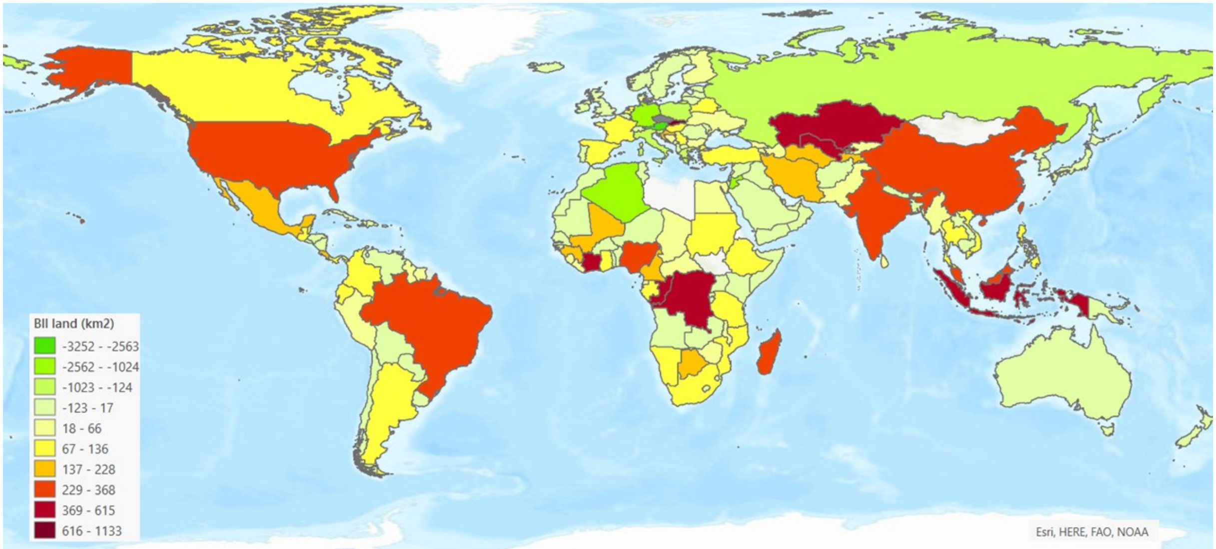

Figure 18 The map of country distribution of the trade balance in the total richness-based biodiversity footprint for the Czech Republic in the years 2001-2005.

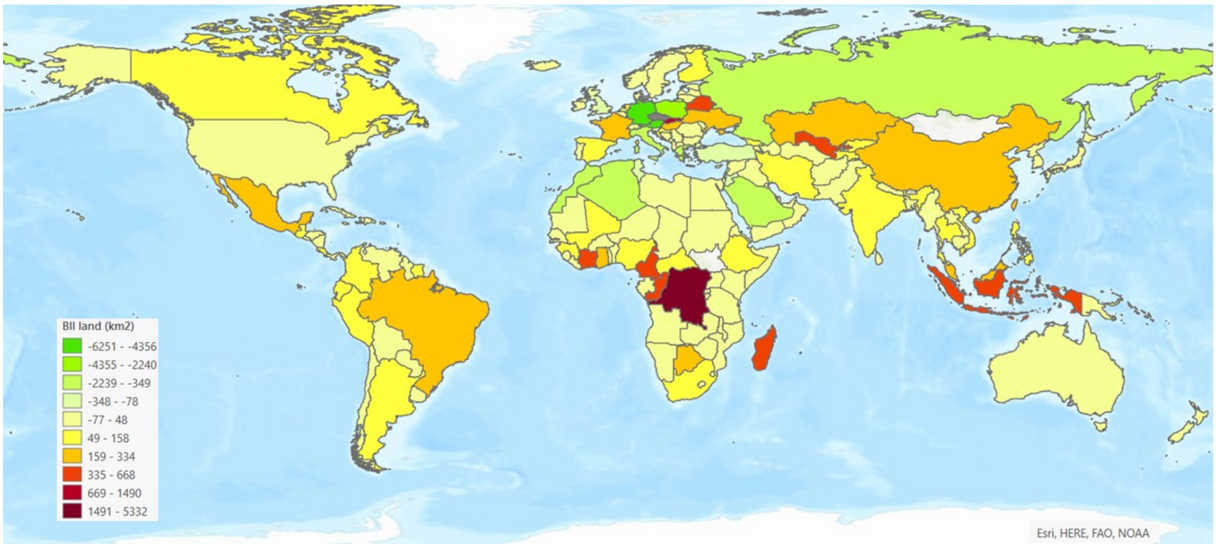

Figure 19 The map of country distribution of the trade balance in the total richness-based biodiversity footprint for the Czech Republic in the years 2006-2010.

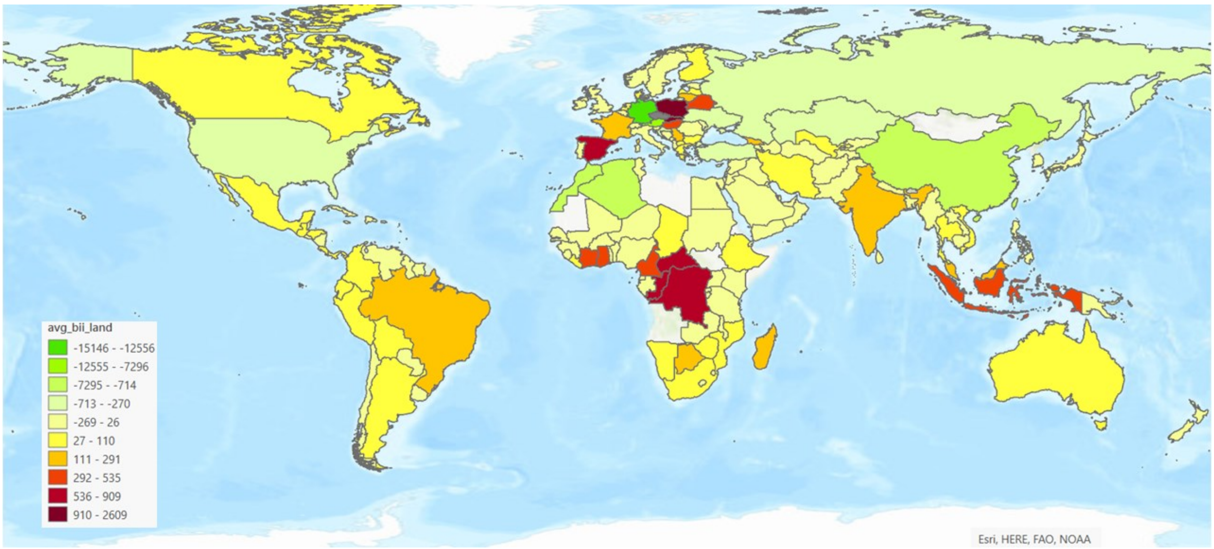

Figure 20 The map of country distribution of the trade balance in the total richness-based biodiversity footprint for the Czech Republic in the years 2011-2011.

5 Discussion

5.1 Limitations and uncertainty

Throughout the text I've repeatedly asserted the data and results are limited in their accuracy. In this section I further summarize and discuss the limitations and uncertainties. I start with the data on trade. Although economic data are commonly treated as hard physical realities, they are actually far from it. Even indices like GDP or national wealth, which (unfortunately) guide national governance, have the character of best available estimates (Piketty 2018). Similarly, national trade statistics, which are the basis of MRIO databases, are approximations based on proxy variables, and their detail and credibility vary among countries. Although authors of MRIO databases go to great lengths to harmonize and cross-validate the underlying data (Tukker, Giljum et al. 2018), a considerable uncertainty remains, especially for "developing" countries. While physical units (mass, volume) are needed for environmental analyzes, trade statistics describe monetary flows. The monetary flows are translated to physical flows based on price factors; nevertheless, prices of commodities are not necessarily globally homogenous and fluctuate in time. One potential bias in responsibility allocation is coupled with the so-called proportionality assumption, which means that imported products are assumed to be distributed proportionally between all end-use sectors in the country (industry, households) (Schulte, Jakobs et al. 2021). The homogeneity assumption affects the accuracy of environmental impact quantification, especially for large countries. It is commonly assumed that the production of traded goods is homogenously distributed throughout the source country (as the trade data scarcely allow further disaggregation). Therefore, the impact intensity factors (e.g., the area of land needed to produce an amount of crop) are based on country averages, which may be far from the actual impact on the location. The issue could be solved by spatially explicit IO approaches (Sun, Tukker et al. 2019), but the results of this study are still limited by this. Overestimation of national footprints of some countries could be coupled with international tourism, when the consumption of foreign citizens (tourists) is added to the domestic consumption (Dietzenbacher, Los et al. 2013). The limitations to the biodiversity data in general were already discussed in Section 2.1 and Section 2.2. Predominantly, there are the limitations coupled with general lack of empirical data and the biases in the available data (Hortal, De Bello et al. 2015). The specific BII data and their limitations were discussed in Section 3.3. The potential bias coupled with the homogeneity assumption also applies to the BII characterization factors I derived. Another limitation is that the characterization factors are static, representing the intactness of ecosystems only in a single year. Although I believe the median BII factors would not change so rapidly for a substantial difference to be apparent, especially considering the uncertainty in the biodiversity data, an ideal biodiversity footprint indicator would illustrate both changes in biodiversity and in the rate of consumption.

Next to those common limitations, there are also problems specific to each land-use category. The homogeneity assumption affects all land uses, but cropland is doubly affected because not only distant locations, but also very different crops are aggregated to derive the impact intensity factors. Thus, although the global IO database is in balance, it might affect the bilateral trade balance, especially if a specific crop with yield (impact) factor far from the average is traded – allocation factors were derived in EXIOBASE to partly mitigate the effect (Stadler, Wood et al. 2018). Forests and pastures are usually used to produce only one good (timber and cattle respectively) – that is neglecting the difference between different kinds of trees and different breeds of cattle -, so those are only affected by the geographical aggregation. On the other hand, unlike croplands which are always anthropogenic, forests and grasslands are originally natural ecosystems which can be exploited by humans to a different degree (from primary forests to artificial plantations). The presence of areas identified as forests or grasslands yet not utilized by humans to produce goods affects the results on two levels. First, it can affect the intensity factors in the MRIO data which were derived as the total output of the product divided by the area needed for its production (for example, total amount of timber divided by the total area of forests in the country). If there is a large area of the land-use category which is not used by people for production – which is common in tropical “developing” countries -, the yield factors can be significantly underestimated, leading to overestimated footprints.[1] On the second level, this affects the BII characterization factors which I calculated as medians of BIIs of all the grid cells where the land-use category is dominant. Unexploited ecosystems are generally coupled with higher biodiversity and intactness. This could lead to overall underestimation of the biodiversity footprint. The authors of the MRIO database and the BII map attempted to fix this bias by removing primary ecosystems and wilderness from the equation, and I removed protected areas when calculating the median BII factors. Nevertheless, the location and extent (and even the definition) of primary ecosystems are disputable, and protected areas do not necessarily correlate with the rate of exploitation; hence, this remains a potential source of uncertainty. On the other hand, the two mechanisms work in opposite directions, and might just cancel each other out.

There is another potential complication for calculating land (biodiversity) footprint of forestry. While the products of cropland or pasture are usually harvested continuously, annually, or once in a couple of years, products of forestry are commonly harvested over a rotation period of many decades. The land (biodiversity) footprint of forest products could be calculated in two ways. The yield factor could be defined as the amount of timber we get if we cut-down a hectare of forest. On a larger scale, this would mean that the products of forestry harvested in one year are attributed just the impacts on the land directly harvested in the year. In the second approach, which is the one used in EXIOBASE, the entire land area needed to sustain annual production of timber is accounted. Therefore, the entire forested area is divided by the annual output of forest products, which results, for example, in a hundred times larger footprint than the direct approach for a forest with 100-year rotation period. Which approach is better is, to a large degree, an ethical question of individual responsibility. I believe the latter approach more realistically captures the context of nature exploitation and the derived results are more precautionary. Nevertheless, larger uncertainty is coupled with the second approach, because the harvest output can greatly differ between years even if the total forest area remains the same (e.g., the timber harvest in the Czech Republic almost doubled between the years 2017 and 2020, see *czso.cz/csu/czso/forestry-2020*), which would result in a very different footprint for the same amount of a product from the same country in different years. Harvest lower than

[1] To provide a simple example, let's assume one hectare of grassland can feed one cow. We have 10 ha of pasture that feed 10 cows. Next to this there are also 2 ha of natural grasslands with flowers (and no cows). The total of 10 cows is divided by the total of 12 ha of grasslands, resulting in yield factor of 0.83 cows/ha. If we then calculate the land footprint of somebody who buys one cow, the consumer is attributed 1.2 ha (using the factor) instead of 1 ha.

proportional to the rotation period would result in footprint overestimation, and vice versa. The aggregation problem for the land-use category "other" is quite obvious, and the uncertainty for this category is arguably the largest. Nevertheless, this land use category contributes the least to the overall conclusions.

With all those sources of uncertainty, which is not realistically quantifiable, it is clear that the results need to be treated as orders of magnitude, rather than accurate measurements. Yet I believe this does not mean the conclusions are not valid. Even the orders of magnitude can provide sufficient indication of emerging patterns and trends, and, indeed, patterns and trends should be the notable outcomes of this study, not the inaccurate numbers.

5.2 Comparison to land footprint

The main rationale (as discussed in Section 2.3) to develop a biodiversity-intactness weighted land footprint is that different land-use categories in different countries are coupled with a different impact on biodiversity. Hence, different land in different countries has different value (here based on biodiversity), and the biodiversity footprint indicator should illustrate this. Does the method devised here fulfill this task? The rate of consumption (trade) plays a major role both in land footprint and biodiversity, and it drives the results of both indicators in the same direction. Nevertheless, an effect of the characterization step is still apparent. As presented in Figure 1, a), the characterization gives relatively greater weight to the products derived from croplands in comparison to products of forestry, which is in line with expectation. Generally, it gives more weight to products coupled with lower BII factors and to imports from countries with lower biodiversity intactness. The characterization step has the greatest effect on the results for trade balance, which is now further affected by which products are traded as well as the difference between the two countries. The relative strength and direction in which the characterization step drives the results for the Czech Republic in the years 2011-2015 is visualized in Figure 21. I took the results of bilateral trade balance for the richness-based biodiversity footprint and for the land footprint (uncharacterized land occupation) and normalized them to the scale from -1 to 1, where the value of 1 is assigned to the country the Czech Republic has the largest surplus with (import larger than export) and the value of -1 is assigned to the country with the largest deficit. Then I subtracted the values for land footprint from the values for biodiversity footprint. In result, the larger the number (in red), the larger weight is given to the import from a country in comparison to export from the Czech Republic. In contrast, when the difference is negative (green) the export from the Czech Republic is given relatively larger weight than the import. This is the combined outcome of the difference in the traded-product portfolio and the difference between the country-specific BII factors. For example, country A imports from country B wheat worth 1000 km^2 in land footprint, and exports timber worth 1000 km^2 to country B. Let's say wheat production in country B is coupled with BII factor of 0.5, and timber production in country A is coupled with BII factor of 0.7. While the trade in land footprint between the two countries appears to be in equilibrium, country A imports biodiversity footprint 571 km^2 larger than it exports. From Figure 21 it is apparent that the Czech Republic imports products coupled with relatively higher impact on biodiversity from many tropical countries like Indonesia, Cameroon, Côte d'Ivoire, Brazil, Madagascar, and also from (non-tropical) Spain, France, Slovakia, or Russia. On the other hand, products coupled with lower biodiversity are exported to Poland, Austria, China, and the Congo. This shows that the characterization step does give the land-footprint results some desired nuance. Nonetheless, the characterization is still limited by the aggregation in broad product categories and across the entire national territory. Adding further detail to the characterization factors would certainly benefit the strength of the indicator in comparison to land footprint.

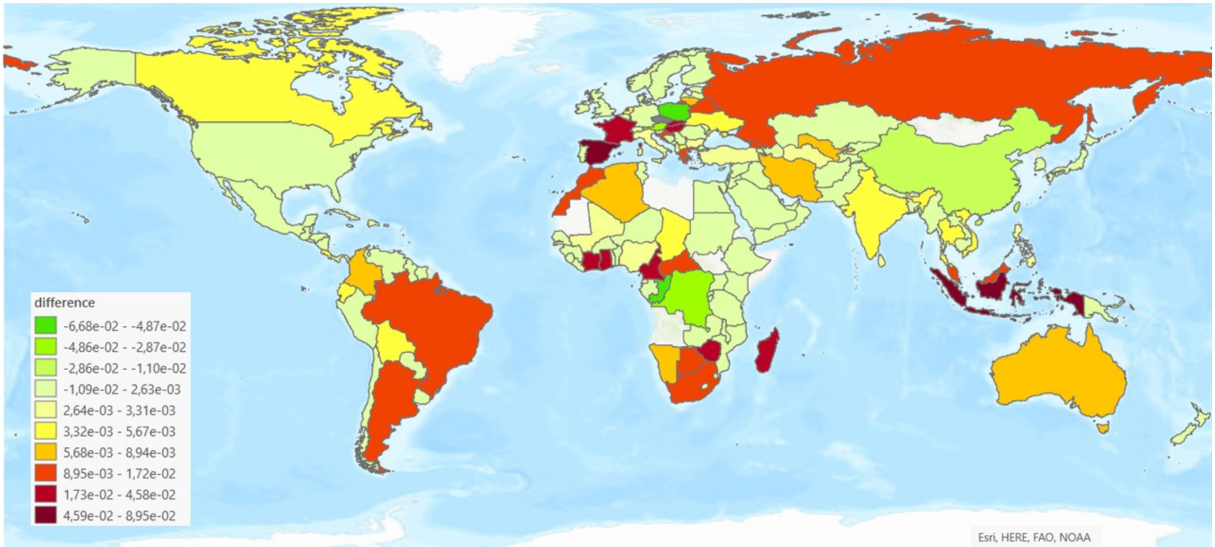

Figure 21 Visualization of the effect of the characterization step. The difference in the bilateral trade balance results for the period 2011-2015 expressed as the richness-based biodiversity footprint and as land footprint, in relative values. Numbers over zero (red) mean that the characterization step moves the balance to the side of surplus (export larger than import) in comparison to land footprint, and vice versa. The light-green and light-yellow indicate there is little difference between land footprint and biodiversity footprint.

5.3 Behavior of the indicator

The accuracy of the results is clearly affected by the quality of the available data. Nevertheless, "*two factors that affect indicator performance are the design of the indicator and the quality of the data that underpin it*" (Collen and Nicholson 2014). The data used here are probably the best currently available, so there only remains to improve the design of the indicator. In this study I developed an approach to biodiversity footprint calculation distinct from others. The nagging question comes up: is it justified to neglect the methods used by others? By neglecting established methods, I crippled any direct comparison of the results to other studies. On the other hand, there is not a one well-established methodology for biodiversity footprint, so the number of potential studies for comparison is very limited anyway, and the available methods are far from perfect, as discussed in Section 2.3. What are the differences in behavior between those indicators and the one developed here?

Any indicator based on an absolute number of extinctions or threats, as the one developed by (Lenzen, Kanemoto et al. 2012), gives greater weight to any impact in naturally more diverse regions and in less "developed" regions, especially the tropics. While this might be the adequate approach to evaluate the global rate of extinction, or to identify impact hotspots (Moran and Kanemoto 2017), in my opinion it is a flawed approach for studies where trade flows are compared. As already mentioned, the design of such indicator inherently leads to the outcome that more diverse ("developing") countries export biodiversity to the less diverse ("developed"). Such conclusion may be interpreted as a critique of exploitative colonial mechanisms persevering in the global order. On the other hand, if used to guide (inter)governmental policy, such indicators could have the opposite effect than envisioned by anti-colonialism, as it would discourage any development that could alleviate poverty in such areas and disrupt global inequality. Another potential effect of such absolute indicators could be that any further or continuing destruction of ecosystems in regions with low biodiversity (namely Europe) appears relatively unproblematic, supporting the voices advocating to restrict any development to those regions (Estrada, Garber et al. 2017). The BII-based method developed here and the approach based on MSA (Hanafiah, Hendriks et al. 2012) are based on relative biodiversity metrics, and thus give the same weight to destruction of nature everywhere. In contrast, Potentially Disappeared Fraction calculated in the LC-Impact method (Verones, Hellweg et al. 2020) – currently the most common method in biodiversity footprint studies – is also a relative measure, but the characterization factors are weighted by factors of presence of endemic species; hence, it also puts greater emphasis on species-rich (or rather endemic-rich) regions. It should be stressed, though, that neither of the methods is intrinsically incorrect. The appropriateness of the design and interpretation of the indicator ultimately goes down to the set research questions, as well as to the underlying values, as discussed in Section 2.1.

The issue of biodiversity is usually framed in terms of loss, decline, extinctions etc., that is a change from one state to another. Biodiversity indicators based on such metrics are therefore indicators of change. While the change of biodiversity is a critical environmental issue, in my opinion it is not the problem biodiversity footprint should, or indeed can, signify. A major attribute of footprint indicators is that they aim to connect the environmental problem to those who are responsible for it, to connect the environmental damage caused by a production to the final consumers of the products who are ultimately responsible. However, attributing responsibility for ecosystem change is problematic. Biodiversity loss caused by transformation of an ecosystem (*land use change*) is a singular act, but farming or forestry (*land use*) are continuous practices. Who is responsible for deforestation? A naïve approach would be to attribute all responsibility to those who personally cut down or burned the trees, or to those who use the timber. But the ultimate goal of such land use change is usually to establish a pasture, plantation, or a cropland to grow other products; thus, consumers of products from those new land-uses should be attributed some responsibility as well. To whom and for how long should the responsibility for this change be allocated? To the consumers of soya grown on newly established croplands? To palm oil produced on plantations established five, ten, or twenty years prior? And what about crop fields established a hundred, five hundred, or thousand years ago? If we divide the responsibility between all the people that bought the products in the past years and will buy the products in the years to come, the responsibility gets diluted to the level of negligibility, especially in the old croplands in "developed" countries. Does it mean that crops grown on such land have no effect on biodiversity and no biodiversity footprint? Such approach would further affirm the neo-colonial dynamics described in the previous paragraph. Furthermore, each product, and thus each consumer, can be attributed only a small, infinitesimal land use change. Yet extinctions emerge only as a cumulative effect of those small changes. Cutting one or two trees does not destroy a forest, only if we cut many of them. This also points to the fact that extinction events are guided by complex, non-linear, and threshold-based dynamics. Is everybody who inflicts a pressure on the ecosystem equally responsible, or only the person who makes the "final blow," who destroys the last refugium? I suggest biodiversity footprint should abandon the framing based on *change* and focus on the *state* of biodiversity. This is what I did in this study by employing the Biodiversity Intactness Index. In fact, this is partly what the authors who employ Mean Species Abundance, among others Hanafiah, Hendriks et al. (2012), do as well. Nevertheless, most authors of biodiversity footprint studies still interpret it in terms of change (disappeared species, extinctions, threats, lost abundance). In my opinion, a more adequate interpretation would be in terms of missing potential species, of species that cannot be present because of the continuous human appropriation of the land. The responsibility for this biodiversity deficit can be coherently attributed to every product extracted from the land. Such interpretation is based on the assumption that the land, if not used by humans, could be restored to the reference state (BII=1). Although this assumption would not always be valid, I believe it is a justifiable simplification.

The design and behavior of any indicator is not a predetermined natural reality we need to discover but a teleological human project. What is the goal of this biodiversity footprint indicator? It is an indicator of the effect human appropriation of land has on biodiversity on the land. The larger the effect, i.e., the lower the biodiversity, the larger the value of the indicator. The approach devised in this study fulfills this condition. Land-uses connected with low biodiversity, e.g., intensive industrial agriculture, have low BIIs and thus high biodiversity footprints, and vice versa. The indicator should also be able to meaningfully display a change, i.e., deterioration or recovery of the ecosystem. Although the detail of data does not allow it in this study, the design of the indicator is adept at this. Deterioration of the ecosystem would be captured by a lower BII, which would result in a larger biodiversity footprint even with the same land area appropriated. Conversely, a reasonable shift to organic agriculture or better forestry practices would be captured by a higher BII, which would result in a smaller biodiversity footprint. Next to consumption reduction, there are two ways to improve the biodiversity footprint score: land-sharing practices increase biodiversity and thus increase the BII, land-sparing practices increase yield and thus decrease land appropriation. The indicator can be used on most scales - local, regional, national, or global (as in this study). There is only the lower limit of the area at which the BII is meaningful, which probably could not be much smaller than the km^2 used here. Of course, more detailed characterization factors (BII) would be necessary for studies at smaller scales. Using richness-based characterization factors, abundance-based, or both can add further nuance to the results, as the abundance-based BII is more related to provisioning of ecosystem services and the richness-based more to their stability.

5.4 Future research

The research project to map tele-coupling of consumption and its distant impacts on the state of biodiversity is far from concluded. There is the general need for more detailed and better-quality data, both for the state of biodiversity and for trade. More specifically, further disaggregated characterization factors, specific for lower geographical levels and for more land uses, would greatly benefit the indicative strength of this biodiversity footprint indicator. For example, the croplands used to grow wheat and vegetables are presumably coupled with different rates of biodiversity intactness, and therefore should be assigned specific characterization factors. However, the applicability of more detailed characterization factors is currently limited by the detail of trade data and the homogeneity assumptions applied therein. Another potential improvement would come from developing dynamic instead of static characterization factors, i.e., BII factors specific for each year rather than using the same characterization factor for all years. Such project is currently limited by the available biodiversity data though. Nevertheless, much can still be done with the method and data employed here. This study uncovered some interesting patterns of biodiversity footprint trade for the Czech Republic, namely the rapid growth in the trade flux, the inter-annual variability, the trade balance evolution etc. The question remains whether those are widespread or isolated patterns among other countries of the world. Several characteristics are potential predictors of the position of a country in global trade in biodiversity footprint, from size or wealth to climatic conditions or geographical position. It would be useful to uncover the effect of other factors than affluence (which is commonly tested), and their relative strength. A similar analysis as in this study should be done for other countries to uncover the trends and patterns typical for them, and to potentially uncover some global patterns.

6 Conclusions

In this study I set to evaluate the role of the Czech Republic in the global dynamics of nature exploitation and ecosystem destruction. Nevertheless, the way this goal is formulated is already burdened by the disparate views on what is nature, the multi-dimensionality of biodiversity, and value-ladenness of the terms "exploitation" and "destruction." The problem I study here can be framed in multiple ways. Therefore, I started by analyzing the Theoretical and contextual background in Section 2. In Section 2.1 I analyzed the different perspectives on biodiversity loss. While Earth's ecosystems are undoubtedly changing, there can be multiple views on what this means. Some people might even claim that the change in global biodiversity is not necessarily a negative phenomenon, but I argue such position is unjustifiable. Yet it is true that different moral frameworks inherently lead to different views on what the problem is and what should we do about it. Furthermore, the state and trends of biodiversity can be depicted differently depending on the scale and dimension we look at. I conclude there is not just one true perspective, but there are multiple legitimate views on this problem. In Section 2.2 I analyze the drivers and trends of biodiversity change. It is mainly the transformation of natural ecosystems and its continuous use (land use and land use change) and overexploitation of natural organisms that leads to biodiversity loss. Nevertheless, factors like the changing climate or invasive species become increasingly damaging. Still, the knowledge about the current state of ecosystems and mechanisms of effect of those drivers and about their interaction is still limited; and hence the options to theoretically predict and model the change in biodiversity are still inadequate. Similar research problems to the one investigated in this study are commonly analyzed using footprint indicators. In Section 2.3 I discussed what those environmental footprint indicators are. Since the environmental problem I'm concerned with here is biodiversity loss, the human impact on biodiversity, the best fitting footprint indicators are land footprint and, especially, biodiversity footprint. I showed there are multiple approaches and methodologies to quantify biodiversity footprint, and I analyzed the advantages and disadvantages of the methods currently used. Based on this analysis I decided to try and develop a novel method to quantify biodiversity footprint that better serves the goal of this study.

The methodology of this new biodiversity footprint as well as the data I used are described in Section 3. I decided to use the Biodiversity Intactness Index, which is a relative metric comparing current biodiversity to a reference "natural" state, as the variable describing the impact on ecosystem. From a global map of BII (Sanchez-Ortiz, Gonzalez et al. 2019) I derived characterization factors that represent the most probable biodiversity intactness coupled with a specific land use category in each country, which I applied to trade data from EXIOBASE 3rx (Bjelle, Többen et al. 2020). The results are presented in Section 4. In Section 4.1 I analyzed the results for the single year 2015, which is a common approach in similar studies. Yet, in Section 4.2, I show on the time series of years 1995-2015 that inter-annual variability can have a great effect on the results and conclusions of such study. Although it remains unexplored whether this is an exception or a common feature among countries, I suggest a better way to analyze such results is in the form of multi-year averages. I also show that the biodiversity footprint imported to the Czech Republic increased nearly six-fold over the analyzed period. In Section 4.3 I showed that biodiversity footprint is imported to the Czech Republic mainly from neighboring European countries, but large flows are also imported from tropical countries like Côte d'Ivoire or Indonesia. In Section 4.4 I showed that the largest imported biodiversity footprint was consistently coupled with products of forestry, followed by vegetables, fruit, nuts category, unspecified crop category, and by meat animals. In Section 4.5 I argue it is not sufficient to analyze only imported products, which would indicate a massive increase of biodiversity impacts of the Czech Republic, but we also need to compare it with exports. I showed that although the imported footprint increased six-fold, the trade balance changed from positive (i.e., larger imports) to negative (i.e., larger exports) at the end of the assessed period. Thus, the trade-balance results show the rapid increase in interconnectedness of the global markets and in complexity of supply-chains. There is apparent a pattern of exploitation of “developing” countries by the “developed” suggested by other studies (Wiedmann and Lenzen 2018), yet it is not too strong and there are notable exceptions. The results of trade balance also show that the imports and exports of products of forestry, although there is the greatest flux, are relatively balanced. The imported biodiversity footprint coupled with vegetables, fruits, and nuts or unspecified crops was significantly higher than the exported. On other hand, the Czech Republic exports significantly larger biodiversity footprint coupled with wheat, cereal grains category, or cattle. In Section 5.1 I evaluated the validity, uncertainty, and limitations of the results. There are multiple sources of uncertainty and of potential bias in the results, hence the results have the character of orders of magnitude rather than of precise measurements. Yet I believe the patterns and trends that emerge from the results, and therefore the conclusions, are valid. In Section 5.2 I showed this biodiversity footprint indicator gives the desired nuance to the results in comparison to undifferentiated land occupation scores. In Section 5.3 I discussed the behavior of the newly developed biodiversity footprint indicator and argued its performance is better, or rather more suitable to the goal of this study, than the currently available methods. Still, there are many research gaps and potential improvements in this field, which are outlined in Section 5.4.

The results clearly indicate that consumption in the Czech Republic has a notable effect on biodiversity in other, often distant countries. The total extraterritorial land necessary to produce the products imported to the Czech Republic in the year 2015 amounted to over 21% of the Czech land area, which is not negligible. On the other hand, even larger area was coupled with the products exported in that year. The maps of the trade balance indicate that most tropical countries, which are hotspots of global biodiversity, export to the Czech Republic more than they import. The same applies for countries like Spain, Slovakia, or Poland though; and, on the other hand, several arguably poorer countries import more than they export. Withal, there are multiple ways to interpret it. Most (neo)liberal economists would probably argue that this is just a manifestation of supply-demand dynamics and a fulfilment of the purpose of open markets to redistribute scarce resources, therefore a positive phenomenon (in ethical sense). In environmental discourse such pattern is commonly interpreted in terms of exploitation, post-colonialism, and outsourcing of impacts – overall in terms of injustice between countries. In my opinion, the pattern in the results of this study is not strong enough to allow such interpretation. The reason might be that the Czech Republic is somewhere in the middle of the exploitation partition. While global inequalities between nations certainly exist, there are also inequalities within nations that might have even greater effect on biodiversity (Hamann, Berry et al. 2018). There is an obvious link between consumption and environmental impacts, and rich people usually consume more, therefore have greater impact on biodiversity (Wiedmann, Lenzen et al. 2020). It seems likely that rich people also consume more foreign goods and therefore contribute more to the extraterritorial biodiversity footprint. Nevertheless, this is just a speculation that cannot be confirmed by the data analyzed in this study. Still, it is clear that there is a significant connection between Czech consumption and impacts on biodiversity in other countries. Through our consumption decisions we affect the state of ecosystems in very distant countries; hence we are also partly responsible for the state of those ecosystems. It is easy to condemn people in "developing" countries for the harm they inflict on so-far natural ecosystems if we already destroyed most natural ecosystems here, in Europe, long ago. However, we, with our consumption, are often the ones who are truly to blame. With the increasing interconnectedness and complexity of international trade it is hard to realize this and maintain a connection with the ecosystems we are dependent on. I hope this study can help to partly elucidate the role we play in the global issue of biodiversity loss.

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