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Land use and deforestation: from cocoa landscapes to global sustainability targets

Doctoral dissertation presented by

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*She reminded me that it is not the land that has been broken,
but our relationship to it.*

Robin Wall Kimmerer

Foreword & Acknowledgments

A doctoral researcher has to accept the sad reality that chances for a thesis manuscript to be read by anyone other than jury members and forced relatives are slim. Except for the acknowledgements. It is therefore tempting to slip in some of the results here, hoping to reach extended circles – but let's leave its role to the summary of Main findings (section 5.1.1 – very convenient read, just saying). Instead, here, I would like to stress some broader aspects of my research that are not reflected in the main text.

I spent over 4 years watching green pixels turn into some other colours. When saying that I research on deforestation and the global land squeeze, the usual question I get is whether it is not a depressing topic. The answer is simple: yes, it is. I spent those years working in a lab with young geographers and climatologists, who, I can tell, do suffer the same curse. Working on a subject which leads you to conclusions such as: 'we are on the brink of massive collapses, suffering and destruction, and what we see today is only the very beginning', while being immersed in a society that clearly seems to find other issues, such as what to call a veggie burger, much more important – this can be, ultimately, unbearable.

I am categorized by my fellow colleagues as an 'activist'. I guess they are right, if being an activist is to try as much as possible to change the trajectory we have embarked on. (Who is 'we', by the way? You have 5 hours). Being an activist while doing science is perceived by some as antithetical – well, you know, you are no longer 'neutral'. But what does it mean to be 'neutral' in the face of a global polycrisis? Is it to remain calm, or is it to scream? Is it to remain observing green pixels disappearing in silence, to report on it via year(s)-long peer-reviewed processes, and gently say 'the forest is dying'? I sense that, in the background, remaining 'neutral' is understood as being apolitical. It means thinking within the system, without questioning axioms such as private property, the free market (free for whom?), and hierarchical institutions.

But this is about politics. This is about what we do and how we decide it.

Over the past few years, I had the opportunity to attend several conferences on forests, climate, and sustainability, where I heard a lot about ways to divert farmers from 'undesirable behaviours', about energy- and technology-intensive systems aimed at making supply chains 'sustainable', or about billionaire multinationals

committing to some new green target. In these venues, certain specific words are barely used, of which one of the most unheard of is ‘capitalism’. This may sound surprising for an economic system that has become virtually hegemonic in a globalised world – but scientists must remain ‘neutral’, what did you expect? The inability or deliberate refusal to call a spade a spade is, in my view, an increasingly worrying problem – it diverts attention away from structural causes to the climate and ecological breakdown (here, I invite you to check section 5.3).

Given the applause that usually follows when someone dares to bring up this subject at these conferences, I can tell that we are many to see that current sustainability approaches are insufficient, and that a more fundamental system change is urgently required. We are many to wonder: What next? What can we do to avoid further delays, further irreversibility, further flirting with tipping points? I spent years wondering, and I gradually discovered a web – made of myriads of individuals, local groups, collectives and organisations – that is already exploring and implementing alternatives, slipping through the cracks, and that, like water turning into ice, has the potential to burst the cap (cliff-hanger for section 5.4).

*

Thank you to all the anonymous game changers, to those perhaps not reading lengthy theses on the state of the world but working to make it a better place.

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To all my colleagues and friends at ELIC, for making the daily trip to Louvain-la-Neuve largely worth-it and for striving to build an inclusive lab. You are the nicest and funniest team – do not forget to unplug from time to time.

To everyone who was part of my previous life in Ghana, where I had the privilege to spend over 3 years learning about the complexities of agricultural supply chains and biodiversity conservation. Meda wo ase. Thank you to Cécile and Mabel, without you, this thesis would not exist.

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Table of Contents

LIST OF FIGURES	11
CHAPTER 1	13
1. INTRODUCTION	13
1.1. <i>The land at the nexus of multiple global crisis</i>	13
1.2. <i>Globalisation and land use displacement</i>	19
1.2.1. Ecologically unequal exchanges	19
1.2.2. Cocoa, a manifestation of land use displacement	21
1.3. <i>Many sustainability initiatives, little effectiveness</i>	21
1.3.1. Sustainability in agri-food systems	22
1.3.2. The global sustainability agenda	31
1.4. <i>Spillovers in complex land systems</i>	35
1.4.1. Indirect land use changes	36
1.4.2. Leakage	36
1.4.3. Rebound effects	37
1.5. <i>Objectives and outline of the thesis</i>	38
CHAPTER 2	41
2. TRANSPARENCY, TRACEABILITY AND DEFORESTATION IN THE IVORIAN COCOA SUPPLY CHAIN	41
<i>Abstract</i>	42
2.1. <i>Introduction</i>	43
2.2. <i>Methods</i>	45
2.2.1. Mapping cocoa sourcing	45
2.2.2. Quantifying the role of cocoa in recent deforestation	47
2.2.3. Linking cocoa deforestation to sourcing	47
2.3. <i>Results and discussion</i>	48
2.3.1. What is the state of traceability and transparency in cocoa sourcing?	48
2.3.2. What is the role of cocoa in recent deforestation?	51
2.3.3. How is cocoa deforestation exposure distributed among actors?	53
2.3.4. Prospects for curbing cocoa-driven deforestation	56
2.4. <i>Conclusion</i>	59
2.5. <i>References</i>	60
CHAPTER 3	65
3. DIRECT AND INDIRECT DEFORESTATION FOR COCOA IN THE TROPICAL MOIST FORESTS OF GHANA	65
<i>Abstract</i>	66
3.1. <i>Introduction</i>	67
3.2. <i>Methods</i>	69
3.2.1. Study area	69
3.2.2. Quantifying direct deforestation and forest degradation	71
3.2.3. Quantifying indirect deforestation and degradation through food crop displacement	73
3.2.4. Validation of assumptions with key informants	78
3.3. <i>Results</i>	78
3.3.1. Direct and indirect drivers of deforestation and degradation	78
3.3.2. Distinct on- and off-reserve deforestation and degradation patterns	81
3.3.3. Scale-dependent patterns	82
3.3.4. Projecting trends in the future	83

3.4.	<i>Discussion</i>	84
3.4.1.	Direct and indirect deforestation and degradation: comparison with other studies and limitations	84
3.4.2.	The role of cocoa expansion in driving food crop deforestation	87
3.4.3.	Is deforestation-free cocoa possible?	90
3.5.	<i>Conclusion</i>	92
3.6.	<i>References</i>	94
CHAPTER 4		99
4.	GLOBAL CLIMATE AND BIODIVERSITY TARGETS ON A LIMITED AND USED PLANET ...	99
	<i>Abstract</i>	100
4.1.	<i>Introduction</i>	101
4.2.	<i>Results</i>	105
4.2.1.	Total land demand for sustainability interventions	105
4.2.2.	Geographic patterns, and increasing pressure on existing land uses	110
4.2.3.	Overlaps between interventions	111
4.2.4.	Unequal land wealth	112
4.2.5.	Suitability trade-offs	113
4.3.	<i>Discussion</i>	114
4.3.1.	Growing land demands on a finite planet	114
4.3.2.	What do these overlaps represent?	115
4.3.3.	The need for breaking silos and centring equity-concerns	116
4.3.4.	How much can these targets be adjusted?	117
4.3.5.	Efficiency	119
4.3.6.	Pressure on land cannot be mitigated without reducing land demands	120
4.3.7.	Limitations	121
4.4.	<i>Conclusion</i>	122
4.5.	<i>Methods</i>	122
4.6.	<i>References</i>	125
CHAPTER 5		135
5.	DISCUSSION AND PERSPECTIVES	135
5.1.	<i>Synthesis</i>	136
5.1.1.	Main findings	136
5.1.2.	Limitations	137
5.2.	<i>Blinders & silver-bullets</i>	140
5.2.1.	Limitations of narrow and shallow approaches	142
5.2.2.	Implications: Breaking silos and diving deeper	146
5.3.	<i>Searching for roots of intertwined problems</i>	146
5.3.1.	Capitalism and the growth paradigm	148
5.3.2.	The dominant worldview	151
5.3.3.	Concentration of wealth and power	154
5.3.4.	Lock-ins and barriers	158
5.4.	<i>Tackling root causes</i>	159
5.4.1.	Navigating thresholds, a degrowth perspective	159
5.4.2.	Societies of care	165
5.4.3.	Implications	166
5.5.	<i>Conclusion</i>	168
CONCLUSION		171
REFERENCES		173

APPENDICES 191

APPENDIX CHAPTER 2 191

APPENDIX CHAPTER 3 214

APPENDIX CHAPTER 4 235

List of figures

Figure 1.1. Feedback loops between climate change, land degradation, biodiversity loss and land scarcity.	16
Figure 1.2. Schematic summary of the scope of chapters 2, 3 and 4.....	38
Figure 2.1. Schematic of the cocoa supply chain in Côte d'Ivoire	45
Figure 2.2. Share of 2019 cocoa sourcing traced to cooperatives, indirect and unknown sourcing.	49
Figure 2.3. Data for the European Union market.....	50
Figure 2.4. Deforestation maps.	52
Figure 2.5. Cocoa deforestation exposure (CDE) in the cocoa supply chain. ..	55
Figure 2.6. Critical departments and protected areas.....	58
Figure 3.1. Study area, datasets and workflow.....	70
Figure 3.2. Land use change definitions.....	77
Figure 3.3. Schematic representation of the expansion of each land use over 2000-2019.	79
Figure 3.4. Deforestation and degradation per driver and per region over 2000-2019.	80
Figure 3.5. Deforestation and degradation per driver off-reserve, in gazetted areas and in the entire high forest zone over 2000-2019.	82
Figure 3.6. Land use in 2019 and projection over 2020-2030.	83
Figure 4.1. Required land area and expansion for the considered interventions and major land uses by 2050 in Mha.....	107
Figure 4.2. Maps of the 2050 spatialised interventions and their overlap with current land uses.....	109
Figure 4.3. Share of current land use areas overlapping with 2050 interventions.	111
Figure 4.4. Percentage of the 2050 interventions overlapped by other interventions in 2050.	112
Figure 4.5. Datasets used and workflow.	123

Chapter 1

Introduction

1.1. The land at the nexus of multiple global crisis

Although nature plays a vital role in sustaining life on Earth, human societies are pushing the planet towards a sixth mass species extinction (IPBES, 2018). Compounded with the ongoing destruction of nature, climate change represents a major threat not only to humans, but to all living beings. The Earth system is entering a period of instability which is already inflicting great harm, and could have disastrous consequences in the near future (Kemp et al., 2022). The most affected populations are those having contributed the least to ecological and climate breakdown, and with the fewest means to adapt to and endure these changes – including small-scale food producers, peasants, pastoralists, Indigenous Peoples, and marginalized groups (IPES-Food, 2024).

In the midst of the environmental turmoil, land takes central stage. Land sustains life. Land and the nature it shelters provide vital contributions to a good quality of life for all, such as the provision of food, fibres, or energy, but also climate regulation, pollination, water purification, or hazard prevention. It provides essential cultural and spiritual benefits, and a sense of belonging (Diaz et al., 2018). Human societies have inhabited and related to the land since the beginning of our times. For more than 10,000 years, the majority of Earth's terrestrial surface has been inhabited and shaped by human use of land (Ellis, 2021). Yet, the rate of global land use change during the past 50 years is unprecedented in human history (IPBES, 2019). Already today, about 25% of the ice-free land surface is converted, and another ~50% has been modified to various degrees (Meyfroidt *et al* 2022).

Currently, major land uses – defined as ‘*the purposes and activities through which people interact with land and terrestrial ecosystems*’ (Meyfroidt et al., 2018) – are (i) grazing land, highly heterogeneous in types and intensity and occupying over a third of the 13,000 million hectares (Mha, ~130 million km²) of global ice-free land surface (~4680 Mha in total, of which ~1480 Mha of intensive grazing); (ii) forestry, gathering plantations and forests used at various intensities, covering about 16% of global land (~1980 Mha), and (iii) cropland, including crops directly consumed by humans and crops for livestock feed, biofuels and other industrial products, amounting to ~1560 Mha, representing 13% of the global land (Matej et al., 2025). Agricultural land thus occupies almost half of the ice-free surface, of which about 80% is used for livestock (Ritchie & Roser, 2019). Though highly influential on land use dynamics, built-up areas and infrastructures represent only about 1% of global land (IPCC, 2022c; Matej et al., 2025). Vegetated land without direct and discernible land use influence accounts for only ~12 to 16% of the ice-free land area, primarily in remote tropical and boreal regions. However, even these remote areas are impacted by climate heating and other global environmental changes (IPCC, 2022a; Meyfroidt et al., 2022).

In the coming years, as human populations grow in size and affluence, **global demands for land and land-derived resources are projected to further escalate**, in particular for food, wood, and urban areas (Weindl et al., 2024). At the same time, proposed strategies for addressing the climate and biodiversity crises, spurred by a growing policy momentum, generate additional demands for land use, such as using more biomass for energy and space to set up wind and solar energy infrastructures, or deploying ‘nature-based solutions’, involving the increase in the protection of remaining natural habitats and the restoration or afforestation of large land areas (or ‘natural climate solutions’ when focused on carbon goals (Griscom et al., 2017)).

Growing demands for land drive changes in land use, making them a major threat to climate, nature and land itself (Figure 1.1, red arrows). Land use change is the major direct driver of terrestrial and freshwater ecosystems loss (IPBES, 2018) and has contributed almost one-third of cumulative anthropic carbon emissions (Friedlingstein et al., 2025). Deforestation has been one of the major manifestations of land use change in recent decades, mainly in the tropics (FAO, 2022), and is one of the largest drivers of greenhouse gas emissions, biodiversity loss, and degradation of nature’s contributions to people (IPBES, 2019). Up to 90-99% of all tropical deforestation worldwide happens primarily in relation with agriculture (Pendrill et al., 2022). The expansion of commercial cropland, pastures and tree plantations alone results in the annual loss of more than 5 million hectares

(Mha) of forests across the tropics and sub-tropics (Pendrill et al., 2019). This positions the food system as a key contributor to climate change, accounting for one-third of global emissions (Crippa et al., 2021) with an increasing trend (M. A. Clark et al., 2020). Changes in land use are not only reflected by area expansions, but also by the intensification of the use of land, extracting more biomass from the same land (Kastner et al., 2022). The intensive management of land can negatively impact biodiversity (Semenchuk et al., 2022), and contribute to climate change, including through the degradation of land¹, negatively impacting on the maintenance of soil carbon sequestration (Ogle et al., 2005).

On the other hand, the climate and ecological breakdown also threatens the land and its capacity to fulfil human demands (Figure 1.1, orange arrows). In particular, land degradation, strengthened by climate change (IPCC, 2022a), undermines the ecological integrity of terrestrial ecosystems and their capacity to, for instance, sustain food production or maintain freshwater and forest resources, and thus reduces the quantity of land suitable for a range of uses (IPCC, 2022a). About 25% of global land area is subject to human-induced degradation (IPCC, 2022a), and about 80% of global arable lands² (Právělie et al., 2021), affecting soil fertility and leading to crop failures in agriculture (IPCC, 2023c). Land degradation directly affects human well-being through the loss of biodiversity and nature contributions to people, which has reached critical levels in many parts of the world (IPBES, 2018).

These pressures on land may create reinforcing feedback loops, such as further expansion spurred by responses to land degradation. It is particularly evidenced by natural – unmanaged – land being converted to agricultural land to sustain production levels (Erb & Gingrich, 2022; IPCC, 2022a). Similarly, land demands for ‘natural climate solutions’ can create vicious circles. As an example, large-scale bioenergy production for direct energy provision or for bioenergy-based carbon capture and storage may intensify, rather than restrain, climate change through indirect land use changes and emissions from land management (Creutzig et al., 2021; Gerssen-Gondelach et al., 2017; Searchinger et al., 2008).

¹ Land degradation is defined by the IPBES as ‘*human-caused processes that drive the decline or loss in biodiversity, ecosystem functions or ecosystem services in any terrestrial and associated aquatic ecosystems*’ (IPBES, 2018). Land degradation is a debated concept; e.g., Indigenous and traditional land use practices, i.e. shifting cultivation and the use of fire for vegetation management, are sometimes considered as leading to degradation. This debate is intertwined with judgments that have long been rooted in deep ethnocentric values and beliefs about ‘modern’ versus ‘backward’ practices. Reflecting these various definitions and uncertainties, estimates of the global extent of land degradation range from 10 to 60 Mkm² (Meyfroidt et al., 2022).

² Meaning that the land is affected by 1 of 5 land degradation processes: aridity, vegetation decline, soil erosion, soil salinization and soil organic carbon loss.

Our inability to phase out fossil fuels significantly and quickly enough could result in additional demand for land for mitigation measures, which could actually worsen climate change.

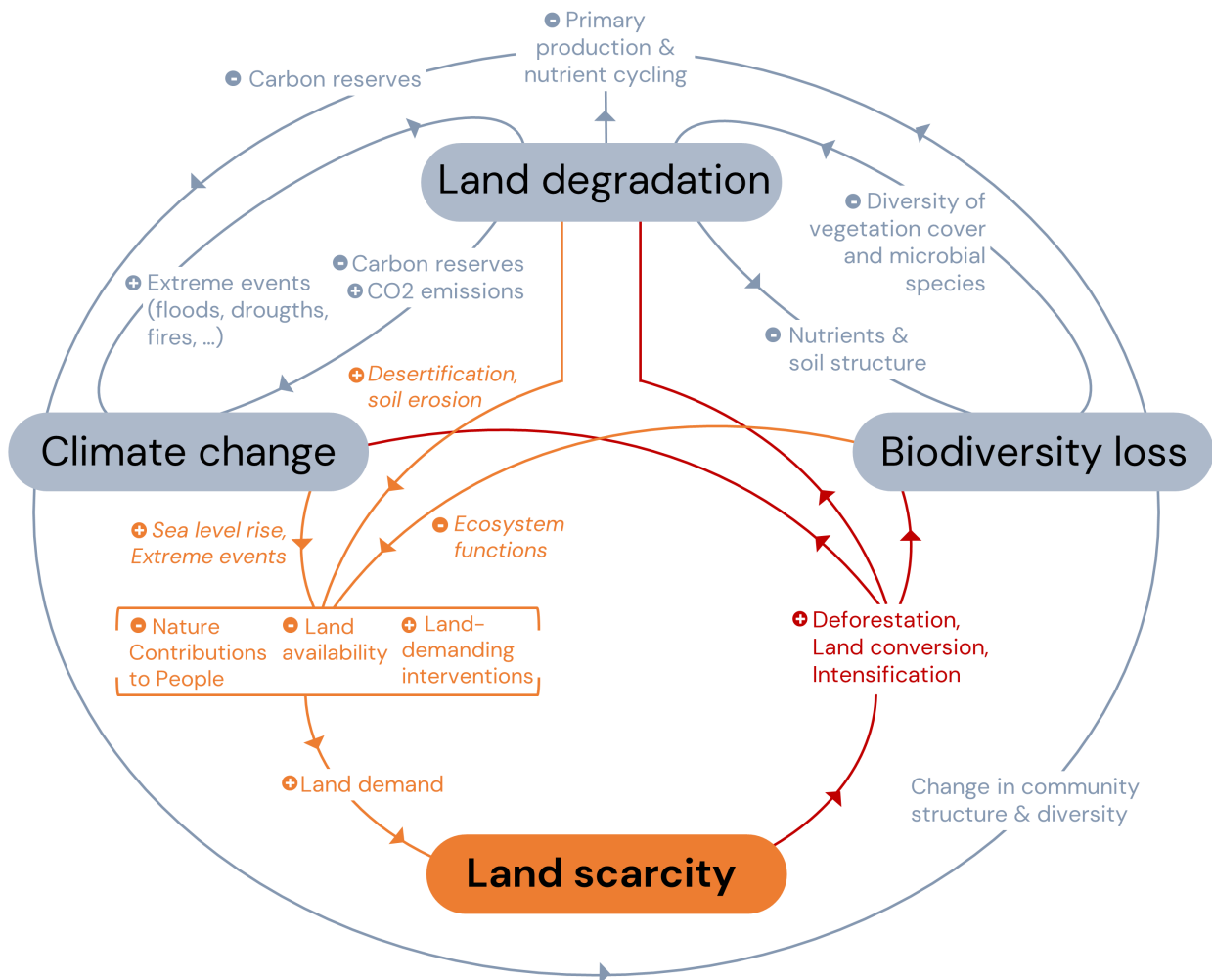


Figure 1.1. Feedback loops between climate change, land degradation, biodiversity loss and land scarcity. The blue loops are a simplification of the Millennium Ecosystem Assessment (2005) schematic linking climate change, biodiversity loss and land degradation³. Here, the orange-red loop was added, highlighting how these three major components of the Earth system breakdown are linked to land scarcity: First, they lead to reduced land availability (e.g., land becoming unusable for agriculture or for habitat) and Nature Contributions to People (NCP) (e.g., reduced crop yields, timber tree mortality). Second, they lead to increased land-demanding interventions (e.g., for bioenergy, renewable energy infrastructures and mining sites, tree plantations, protected areas) – via specific mechanisms, of which non-exhaustive examples are provided in italic (e.g., sea-level rise, desertification, degraded ecosystem functions). Together, these two sets of processes contribute to further increasing the global land scarcity. This scarcity and pressure on land leads to further deforestation, land conversion and intensification, themselves further increasing biodiversity loss, land degradation and climate change.

³ Soil erosion and desertification in the MA schematic were changed here to land degradation.

On our limited and already used planet, growing demands for land means greater competition for land, which entails manifold trade-offs. Many of these land demands require high-quality, productive lands and rich soils, or access to infrastructures, meaning that different actors and land uses often compete for the same land (Abeygunawardane et al., 2022; Brunelle & Makowski, 2020). At the same time, very little land is potentially available while being ‘empty’ or ‘free’ of trade-offs (Lambin et al., 2013; Meyfroidt et al., 2022). Sometimes unnoticed, land provides functions, benefits and services – which can differ for people across and within societies who attach different meanings and values to land (Ellis et al., 2019). Any changes in land use have thus the potential to alter these functions hence generating trade-offs (Ellis et al., 2019; Foley et al., 2005; Niewöhner et al., 2016). Trade-offs can occur between people or places (Marques et al., 2019), or between global and local issues (Poudyal et al., 2018) or current and future outcomes, such as between meeting immediate human needs and maintaining the capacity of the biosphere to provide goods and services in the long run (Foley et al., 2005). Trade-offs between nature conservation and food production are common (Meyfroidt, 2018), and the increasing implementation of large-scale ‘nature-based solutions’ will further exacerbate existing trade-offs and generate new ones (Fleischman et al., 2022).

Humanity is facing an unprecedented mismatch between growing multiple land demands on one side, and the concurrent declining availability of land and its ability to provide nature's contributions to people. Concepts such as ‘land squeeze’ (IPES-Food, 2024; Searchinger et al., 2023), ‘land crunch’ (King et al., 2023), ‘land gap’ (Dooley et al., 2022), ‘land scarcity’ (Lambin & Meyfroidt, 2011) or ‘land squeeze’ (IPES-Food, 2024) denote this anticipated situation of enhanced land competition, which entails complex social dynamics and dimensions of justice; e.g., land concentration leading to the loss of farmers’ and communities’ control over their land (IPES-Food, 2024). These concepts have emerged in recent years and have helped in increasingly recognizing land as being at the nexus of sustainability issues (Erb et al., 2024).

Many people, governments, companies, civil society organizations, scientists, have been working and acting on these issues for long. **Multiple forms of interventions have started to be deployed**, such as sustainable sourcing initiatives aiming at reaching ‘zero deforestation’ land-based commodity supply chains, the development of renewable energy, carbon capture through tree planting or bioenergy crops, and nature-based solutions and the related commitments to protect 30% of our land and restore 30% of degraded ecosystems. Socio-economic and cultural

pathways may lead different people to focus on and believe in particular interventions, sometimes perceived and portrayed as silver-bullet solutions.

Yet, these interventions face complexities. First, as introduced above, they themselves add to the land demand and pressures, possibly targeting and competing for the same land. Second, land systems are complex systems, so interventions rarely result in just one single outcome, instead, they often cause complex reactions, feedbacks, and spillovers (Meyfroidt et al., 2018). Third, interventions targeting a given sector or focusing on proximate causes might be ineffective to address more structural problems driving the land squeeze (Garrett et al., 2018, 2025).

In this thesis, I take a critical perspective⁴ on how sustainability interventions face these complexities. The objective is to bring an improved understanding of the land demand outcomes and limitations of current sustainability interventions, and to explore how to improve approaches to sustainability. Each chapter broadens the studied geographic scale and the assessed interventions, land uses, and impacts (Figure 1.2). **Chapter 2** starts at the *individual supply chain level* with company-led traceability as a tool to ensure deforestation-free commodity sourcing. Then, moving to a broader scale, **chapter 3** assesses at the *sector level* zero-deforestation commitments aimed at eliminating deforestation from the entire commodity sector in specific landscapes. These two chapters focused on the specific context of cocoa production and supply chain in the two major producing countries, Côte d'Ivoire and Ghana, respectively, and discussed the limitations of these interventions in curbing deforestation. Next, at the *global-level*, **chapter 4** focuses on major land-based interventions considered within the global sustainability agenda (e.g. conservation, restoration, renewable energy) aimed at addressing the biodiversity and climate crisis, and assesses their implications on land. Expanding from these learnings, **chapter 5** explores transformative changes required to move beyond current conventional approaches to sustainability, considered as 'narrow and shallow'.

In this introductory chapter, I first provide some background, starting, in section 1.2, by how globalization has shaped an interconnected world, and has driven the displacement of land uses across the globe, generating distant impacts, such as deforestation in cocoa producing countries to produce chocolate for remote companies and consumers (the focus of **chapters 2 and 3**), and land being devoted to meet specific sustainability targets defined at national or international level, often

⁴ Here, *critical perspective* is used in the broad sense of *critical thinking* and not from a *Critical Theory* standpoint – which centres on analysing and challenging systemic power relations in society and lies in the realm of philosophy and social science. Chapters 2-4 focus instead on empirical data-driven and quantitative analyses and do not concentrate on dynamics of oppression. Yet, chapter 5 does so more explicitly, with a clear aim to transform society – thus bringing the thesis closer to the strand of critical geography.

away from local communities (the focus of **chapter 4**). Section 1.3 presents sustainability initiatives in the agri-food system and the cocoa sector, as well as the interventions considered as part of the global sustainability agenda, and their limitations and implications on land and people. Lastly, section 1.4 dives into land system dynamics, particularly policy leakages and other processes which render these dynamics intrinsically complex, ultimately making silver-bullet solutions inadequate. I finally further develop the objectives and outline of this thesis (section 1.5).

1.2. Globalisation and land use displacement

Today's world, shaped by capitalism, liberalization and globalization, is strongly interconnected through global and numerous webs of connections, made of networks of telecommunication, pipelines, electricity grids, roads, canals, and air and shipping routes, supply chains and trade, finance, and monetary systems, but also of links among climate, ecological, and land systems⁵ (Lawrence et al., 2024). International trade has grown significantly over the past decades (Traverso and Schiavo 2020, IMF 2023). As Lawrence et al. 2025 point out: *'the total value of world merchandise trade increased twelve-fold between 1980 and 2022 to nearly 25 trillion US dollars annually (at current prices), and container port traffic has more than tripled since 2000 to almost 800 million 20-foot-equivalent-units in 2020'*.

As a result, we live in a world where local decisions can impact distant locations. Socio-ecological systems across the world are more or less interconnected, and a system in one region can affect one or multiple other system(s) elsewhere – these distal linkages, or *telecouplings* (Eakin et al., 2014; Liu et al., 2013, 2018), have far-reaching socioeconomic and environmental impacts over distances. One of these impacts is the displacement of land use, often coming along with ecologically unequal exchanges.

1.2.1. Ecologically unequal exchanges

Urbanization, trade liberalization and the growth and consolidation of transnational corporations have caused increasing spatial separation of the locations of

⁵ Land systems are *'terrestrial social-ecological systems where human and environmental systems interact through land use'* (Meyfroidt et al., 2022), encompassing the various actors and demands for land, the technologies, institutions, and cultural practices shaping land uses, and the interrelations and dynamics between land uses and the environment (Meyfroidt et al., 2018; Verburg et al., 2015).

production and consumption of products (Garrett & Rueda, 2019; Lambin & Meyfroidt, 2011).

Akin to *virtual water*, i.e. the volume of water embodied in traded goods, *virtual land* or ***embodied land*** refers to productive areas hidden in imported or exported goods, and is equivalent to a transfer of land between countries (Erb et al., 2009, p. 200; Würtenberger et al., 2006). From a national standpoint, the country size of importers is being expanded by an area located outside the country's borders and used for the production of food, fiber, timber, minerals, etc. (i.e. displaced land use or appropriated land) (Würtenberger et al., 2006). On the other hand, exporters use shares of their land for the production of products consumed abroad (i.e. absorbed land use) (Li et al., 2024; Meyfroidt et al., 2010).

These ***land use displacements*** – i.e. ‘*a temporal, spatial, social or sectoral separation between consumption and production of a material good*’ (Meyfroidt & Lambin, 2009), or, in a broader sense, ‘*a geographical shift of land use from one place to another*’ (Meyfroidt et al., 2013) – generate complex cascading socio-economic and environmental impacts. The displacement of land use can lead to land use change and deforestation (Pendrill et al., 2019), biodiversity loss (Marques et al., 2019), and many other environmental externalities (e.g. pollution, wastes, water overuse) (Shandra et al., 2008) in producing countries. Importing land use thus equates to exporting ecological impacts (Mayer et al., 2005). Many agricultural commodities flow from the Global South to the Global North resulting in the shifting of ecological burdens from high-income to lower income countries through trade (Li et al., 2024), though it also happens within and among Global South or Global North countries. This is captured by the concept of ***ecologically unequal exchanges***⁶ (Givens et al., 2019). Land use displacement, facilitated by power imbalances, thus further increases social and environmental inequalities (Chen et al., 2021; Hickel et al., 2022).

Cocoa production and global sustainability interventions require vast tracts of land, and imply unequal exchanges, already generating complex impacts across telecoupled systems. Below, I provide more background on cocoa and its impact in Côte d’Ivoire and Ghana, while implications of global sustainability interventions are developed in section 1.3.2.2.

⁶ Citing Givens et al. (2019) at length: ‘*According to this theory, global political–economic factors, especially the structure of international trade, shape the unequal distribution of environmental harms and human development; wealthier and more powerful Global North nations have disproportionate access to both natural resources and sink capacity for waste in Global South nations.*’

1.2.2. Cocoa, a manifestation of land use displacement

Cocoa is embedded in a long colonial history, in which the production of luxury commodities such as sugar, tea, coffee, cocoa, banana, palm oil or peanuts, outsourced to tropical colonies, has been causing unprecedented environmental harm and pervasive subjugation of people (McMichael, 2009). Cocoa, mainly consumed in high-income countries, thus typifies the displacement of land use from rich to poorer regions of the world and the associated shift of ecological and social burdens.

Within the last decades, the cocoa industry grew steeply. Cocoa production is one of the most important economic sectors in Western Africa, employing millions of smallholder farmers and workers. Though confronted with drops in production over the last years, Côte d'Ivoire and Ghana are the world's largest producers of cocoa, with one million smallholder farmers producing annually over 2 million tonnes of cocoa beans, i.e. about 40% of the world's cocoa harvest (FAO 2020), in Côte d'Ivoire, and over 800,000 smallholder farmers producing ~800,000 tonnes of cocoa beans yearly (FAO 2022), i.e. ~20% of global cocoa production, in Ghana. Yet, cocoa's key contribution to the Ivorian and Ghanaian economies has gone along with massive deforestation to make space for an ever-growing sector, particularly seeking for fertile forest soils (Ruf, 1987).

Chapters 2 and 3 focus on deforestation, one of the most acute environmental impacts of the international demand for cocoa. Section 1.3.1.3 describes how actors of the cocoa sector are attempting to address this issue.

1.3. Many sustainability initiatives, little effectiveness

Here, I shortly sketch the vast landscape of 'sustainability' initiatives, made of diversified – sometimes opposite – approaches which have evolved rapidly in recent history. I first explore initiatives aiming at addressing deforestation in the agri-food sector, with a focus on traceability and zero-deforestation commitments in the cocoa sector (for **chapters 2 and 3**), before peering into major interventions considered within the global sustainability agenda (for **chapter 4**).

It is important to stress that *sustainability* is a normative concept, its meaning varying across people and cultures. At its heart lays the question: 'What is it that we want to sustain?'. The United Nations conference on the Human Environment of 1972 marks the point where sustainability and sustainable development became synonyms (Biely, 2020). The following Brundtland Report of 1987 defined sustainable development as '*the development that meets the needs of the present*

without compromising the ability of future generations to meet their own needs.' This in turn begs the question: 'What is it that qualifies as a need?'. Without delving into this question, let us just say that most sustainability definitions relate to the aim of balancing the typical three dimensions of environmental, social and economic spheres. Yet, what this concretely means remains contested (Lozano, 2008), as is exemplified by the weak vs. strong sustainability divide. While *weak sustainability* depicts sustainability as the overlap between social, economic, and environmental spheres, *strong sustainability* brings the economic sphere out of its vacuum and place it within the natural environment – depicting the economy embedded within the social system, which is embedded in the natural system (Biely, 2020; Lozano, 2008). The enmeshment of sustainability and development was finally completed by the instalment of the UN Sustainable Development Goals (SDGs) as a normative agenda outlining an operational vision of what is understood by sustainable development.

1.3.1. Sustainability in agri-food systems

Here, I focus primarily on policies related to forests, which are the focus of the chapters 2-3, though many other sustainability issues, especially social impacts, are entrenched in global agri-food systems. I focus on describing these policies⁷, but do not exhaustively review their effectiveness, efficiency or equity (see e.g., Furumo and Lambin, 2021; Garrett et al., 2018, 2019, 2021; Lambin et al., 2018).

1.3.1.1. The rise of supply chain sustainability initiatives

Prior to the mid-1990s, governance of land use was primarily the domain of governments who were setting constraints on land use in their jurisdiction, encouraged by international conventions (Cashore, 2002). With the acceleration of globalization and neo-liberal agendas of deregulation, governance shifted from being place-based to flow-based (Sikor et al., 2013), and from state- to non-state and market-driven governance (Cashore, 2002). In the 2000's, NGOs' enormous pressure on agribusiness companies shifted attention from governments to the private actors (Bager, 2022), deemed easier to influence than domestic- and international policy networks (Cashore, 2002). Indeed, only a small number of traders handle most of the world's trade in each of the main land use-demanding commodities and this market concentration is thus perceived as providing transnational corporations

⁷ For a timeline highlighting events of relevance to governance of agro-food commodity supply chains, see p.68 in Bager (2022).

with the leverage to improve the sustainability of a large proportion of the world's sourcing (Folke et al., 2019; Grabs et al., 2024).

As a result, several private and public-private governance interventions begun to emerge in the 2000s particularly through a range of voluntary **supply chain sustainability initiatives** (SSI)⁸. Such mechanisms can be grouped into four main categories (Lambin et al., 2018): (i) **multi-stakeholder aspirations**, such as the Forest Positive initiative of the Consumer Goods Forum, the Tropical Forest Alliance 2020, the 2014 New York Declaration on Forests, and the 2015 Amsterdam Declarations; (ii) **company pledges** such as zero-deforestation commitments (ZDCs) signalling a company's intention to eliminate deforestation from its supply chain (Garrett et al., 2019; Lambin et al., 2018) – many actors at all levels of forest-risk supply chains from production to retail have adopted ZDCs (Grabs et al., 2021); (iii) **company codes of conduct** such as Unilever's Responsible Sourcing Policy; and (iv) **sectoral standards** across major forest-risk commodities including incentive-based mechanisms (e.g., the Rainforest Alliance, or those developed by roundtables⁹), and sanction-based mechanisms such as moratoria on purchases from suppliers that do not comply with standards (e.g. the Soy and the Beef Moratoria).

In parallel, in the 2000s, various **subnational approaches** to governing land use emerged: REDD+¹⁰ projects to incentivize subnational action on climate change and biodiversity loss, and then landscape or jurisdictional approaches, which started taking hold from 2015 (Bager, 2022). Landscape and jurisdictional approaches are governance initiatives promoting sustainable resource use in a specific landscape or jurisdiction through a formalised collaboration between all stakeholders (Reed et al., 2016; von Essen & Lambin, 2021). These approaches based on pre-competitive multi-stakeholder collaboration between companies and public actors, at the landscape level, are seen as having the potential to deliver consistent impact at scale (Bakhtary et al., 2020; Carodenuto, 2019; IDEF, 2021), and to internalise indirect sourcing (cf. section 1.3.1.4) and deforestation drivers beyond the focal commodity (cf. section 1.4.1) (zu Ermgassen et al., 2022).

A **legality approach** also started in the timber sector, focused primarily on avoiding illegal activities (Bager, 2022), which started merging with the market-driven

⁸ For a nomenclature of initiatives in the voluntary environmental governance literature, see table S1 in Garrett et al. (2021) [here](#).

⁹ Such as the Roundtable on Sustainable Palm Oil (RSPO), the Forestry Stewardship Council (FSC), the Program for the Endorsement of Forest Certification (PEFC), the Round Table for Responsible Soy (RTRS), the Leather Working Group (LWG) and the Global Roundtable for Sustainable Beef (GRSB).

¹⁰ Reducing Emissions from Deforestation and Forest Degradation.

governance approaches in the form of **supply chain oriented public policies**, signalling a ‘come back’ of public governance, with several countries including France and Germany passing due diligence legislations (Bager, 2022). This eventually led to the **European Union’s Regulation on Deforestation-free Products** (EUDR) requiring companies importing deforestation-risk products (cocoa, coffee, beef, leather, soybeans, palm oil, wood, and rubber, and their derived products such as chocolate) to demonstrate that imports do not originate from land deforested or degraded after the cut-off date of December 31, 2020 (EU, 2023). One of the regulation's main tenets is the requirement for traceability at the plot level, which has sparked accelerated efforts among businesses to map the farms of their supply base.

1.3.1.2. Traceability, transparency and accountability

Traceability, according to the Accountability Framework initiative (AFi), is ‘*the ability to follow a product or its components through stages of the supply chain (e.g., production, processing, manufacturing, and distribution)*’, and **transparency** is ‘*the public disclosure of information in a form and manner accessible to stakeholders*’ (AFi, 2025), referring to information not only about the company’s supplier network, but also about the business, policies, activities, risks, performance, impacts, and other relevant attributes (AFi, 2025). Transparency, as it is currently understood and used in supply chains, takes various forms and does not necessarily involve making all data available to everyone, especially competitive or proprietary information. Data sharing is adapted to the recipients, e.g. independent monitoring organisations vs. the public (Dontenville et al., 2022). Supply chain **accountability** includes mechanisms such as verification, audit, complaints mechanisms that ‘*demonstrate a willingness to make oneself ready to answer for one’s acts*’ (Dontenville et al., 2022). This may include acknowledging mistakes and unintended negative consequences, as well as the steps taken to remedy them (Dontenville et al., 2022).

With pressure building up on companies – mainly from consumers and NGOs – (Bager, 2022), there has been a growing interest in supply chain transparency in recent years, in particular ‘public transparency’, i.e. making direct information on the sustainability of production processes and commodity characteristics available to the wider public (Gardner et al., 2019; Mol, 2015). Transparency has a strong outward-looking dimension of performance demonstration and trust-building (Dontenville et al., 2022), and plays a central role in the accountability of companies, as it can be critical to their credibility (Gardner et al., 2019). Actors such as AFi, which provides unified guidance to companies on reporting and disclosure

practices, and Trase, which maps supply chain flows of major deforestation-risk commodities, participated in shaping this drive towards traceable and transparent supply chains by taking concrete steps to provide knowledge and transparency on often opaque production practices (Godar et al., 2016). Most commodity sectors however still depend on voluntary measures to ensure that disclosures happen – underscoring the continued lack of transparency in global supply chains (Gardner et al., 2019).

Importantly, traceability and transparency raise power and equity concerns: e.g., who owns the supply chain data, who has access to it, who bears the related costs? In both company-led and state-led traceability systems, physical coordinates, bio-data and data on farming households are registered but not made available to the public, further concentrating data and associated knowledge and power in private hands or central states. These supply chain policies are often unilateral initiatives stemming from northern companies and governments, such as the EUDR – with implications in the three equity dimensions: procedural (i.e. equity of decision-making processes); distributive (i.e. distribution of costs and benefits); and recognition (i.e. whose norms, cultures and knowledge systems are recognized and empowered) (McDermott et al., 2025). Traceability systems are for instance built on a conception of privately owned and managed, and georeferenced farms, hence overshadowing traditional tenure systems (McDermott et al., 2025). How ‘forest’ is defined also plays a critical role; e.g., the EUDR uses FAO’s definition¹¹ which is more expansive than the Ghanaian¹² and Ivorian¹³ law – farmers holding legal and traditional rights on land considered as forest by the EUDR will lose the possibility to sell cocoa from those areas to the EU (McDermott et al., 2025). Unless properly co-designed and implemented, such policies risk further entrenching inequities, as developed in chapter 5.

1.3.1.3. Sustainability in the cocoa sector

As a result of increased civil society pressure, the cocoa industry gradually started engaging with sustainability. Two major events, not related to deforestation, initiated **industry-level initiatives** in the 2000s (Thorlakson, 2018): (i) major pest and disease outbreaks around the turn of the 1990’s led to the launch of the World Cocoa Foundation in 2000, with an early focus on productivity to ensure the long-

¹¹ i.e., >0.5 hectare with trees >5 metres high and a canopy cover of >10%, or trees capable of reaching these thresholds in situ (EFI, 2025).

¹² i.e. ≥1 hectare with trees ≥5 metres high and a canopy cover of ≥15% or trees capable of reaching these thresholds in situ (EFI, 2025).

¹³ i.e. ≥0.1 hectare with trees ≥5 metres high and a canopy cover of ≥30% or trees capable of reaching these thresholds in situ (MINEF, 2019).

term sustainability of cocoa production and avoid supply shortages, and (ii) confronted to scandals related to child labour, the industry established the Harkin–Engel protocol in 2001, committing companies to stop the worst forms of child and forced labour in cocoa production by 2005 (which eventually failed).

In the turn of the 2010s, after a decade of heightened lobbying by NGOs and certifiers, chocolate companies made major **third-party certifications** commitments, aiming at sourcing 100% of their cocoa from Fairtrade, Rainforest Alliance, or UTZ certified producers. In the course of the 2010s, however, certification lost its role as the ‘guiding light’ in the cocoa sustainability space. Companies were faced with high costs and low returns; certification gave mixed results, particularly with regard to child labour, and therefore did not protect their brand – while also not proving to be a sales driver (Thorlakson, 2018). Consequently, large companies with sufficient resources started launching their **own in-house sustainability commitments, programs and standards**, either in conjunction with certification or replacing it (Ingram et al., 2018; Thorlakson, 2018). This shift made manufacturers more reliant on traders to build and maintain strong relationships with producer groups, spurring efforts in increasing traceability in these forest-risk supply chains. Traders such as Barry Callebaut and Cargill hence started getting in on the act – with sustainability seen as a major differentiation opportunity for them (Thorlakson, 2018).

Many companies with forest-focused supply chain policies¹⁴ (FSPs) have zero net or gross deforestation commitments, though, in practice, they tend to limit their efforts to not purchasing cocoa from legally protected zones¹⁵ (Addoah et al., 2025). Companies’ FSPs focus largely on agroforestry, often conflating it with forest restoration or considering it as a deforestation-related activity (Addoah et al., 2025). FSPs activities concentrate mainly on shade tree seedlings distribution. More recently, a few buyers introduced performance payments for tree planting and survival (Addoah et al., 2025).

In 2017, the **cocoa and forest initiative** (CFI) was launched by a consortium of 35 chocolate and cocoa companies together with the governments of Ghana and Côte d’Ivoire, aiming to contribute to ending deforestation and forest degradation in the cocoa supply chain. It includes a commitment by companies not to source cocoa produced on forest land that was converted to agriculture post-2018. This

¹⁴ ‘Forest-focused supply chain policies (FSPs)’ are formal and/or public declarations about how forest impacts are considered within a company’s supply chain, often also including statements about social safeguards and livelihood targets (Garrett et al., 2021).

¹⁵ A form of market exclusion mechanism (excluding suppliers based on undesirable production practices such as deforestation).

collective commitment to zero deforestation cocoa exemplifies the *own supply chain era* in sustainable sourcing practices (Carodenuto & Buluran, 2021; Thorlakson, 2018).

Subsequently, the CFI, together with REDD+, and the recently revised Ivorian Forest Code¹⁶, supported the launch of **landscape level initiatives** in Ghana and Côte d'Ivoire in recent years, though they are yet to fully materialize and show impact (Addoah et al., 2025).

Most recently, the **African Regional Standard** for sustainable and traceable Cocoa (ARS-1000)¹⁷, aiming at enhancing sustainability, traceability, and quality in cocoa production across the region, was developed under the Ivorian and Ghanaian governments' impulse. In spite of great overlaps with the EUDR, some actors highlight their mismatches (e.g., cut-off date) (EFI, 2024; IDH, 2023) which may complicate understanding on the ground, and call for harmonization (IDH, 2023). When operationalized, this standard is meant to be mandatory for all cocoa supply chain actors in both Côte d'Ivoire and Ghana – covering the entire sector in these two major producing countries, with potential impact at scale, given it is effectively implemented.

Despite these developments, traders' public sustainability commitments, which are mainly adopted by large transnational traders, remain limited in coverage; in 2018, only a fourth of the global cocoa was covered (Parra-Paitan et al., 2023). This share would increase up to 60% if the indirect sourcing of these large traders was included in their sustainability commitments (Parra-Paitan et al., 2023). The **EUDR** and the ARS-1000 represent thus a major change, from voluntary initiatives remaining limited in scope and, often, in effectiveness (cf. section 1.3.1.5), to mandatory practices imposed to all actors (in the case of the EUDR, to all companies importing the forest-risk goods covered by the legislation in the EU). The foreseen enforcement of the EUDR by the end of 2024, then postponed to 2025 (presuming that no further major rollback happens), has launched a race for companies to trace and measure compliance at the farm-scale, which has concentrated much attention and resources.

¹⁶ Which reclassified degraded forest reserves into agro-forests meant to be managed by non-state actors (Law n 2019-675).

¹⁷ Key ARS-1000 guidelines regarding deforestation¹⁷ include: (i) no deforestation and degradation of primary forests (from June 2021); (ii) no deforestation or degradation of secondary forest, unless a legal land title, landowner permission and/or customary land rights, and government permits are available; (iii) no farming in protected area, unless there are explicit permissions under national law; (iv) national traceability systems from farm to export are required (cf. section 1.3.1.4) (ARSO, 2021).

1.3.1.4. Cocoa traceability, the new sustainability token

Today's consumers would expect companies to know the origin of the products they sell – yet, global agri-food supply chains remain opaque, preventing linking supply chains of specific companies to associated sustainability risks (Gardner et al., 2019; zu Ermgassen et al., 2022). Using global cocoa trade data of 2018, Parra-Paitan et al. (2023) showed that only about a fifth of worldwide cocoa companies, handling a third of global cocoa exports, reported being able to trace their cocoa back to farmer cooperatives. This underscores the challenge posed by the EUDR, requiring plot-level traceability. Further, only 8% of cocoa companies (handling 23% of cocoa exports) were transparent about the identity or location of their direct suppliers (Parra-Paitan et al., 2023), highlighting how transparency lags behind traceability.

Paving the way to the EUDR, the CFI has one of its key objectives aiming at reaching 100% traceability (initially intended for 2022 (WCF, 2019)), but this goal was confined to the *direct* supply chains of signatories (CFI, 2021). Similar to many other contexts, supply chains are complex and include so-called *indirect sourcing* (zu Ermgassen et al., 2022). The supply chain in Côte d'Ivoire is structured around two major flows: (i) farmer cooperatives gathering their members' production to honour their sales contracts with traders, (ii) individual buyers, i.e. *traitants*, collecting cocoa from many local intermediaries, i.e. *pisteurs*. Traders can buy directly from cooperatives and licensed buyers (*acheteurs agréés*), or indirectly from other local intermediaries, before exporting to various countries (Nitidae and EFI 2021). What the sector calls *direct sourcing* is thus traceable to the first buyer, cooperatives in most cases, but not necessarily to farm. Consequently, the 100% direct sourcing traceability target can have very different meaning depending on the proportion of direct versus indirect sourcing – and this information is not publicly available. In addition, it is challenging to assess where companies source their cocoa from, let alone monitor their progress in implementing sustainable sourcing initiatives (Dontenville et al 2022). **Chapter 2** aimed at addressing this gap, by shedding light on the state of traceability and transparency in the cocoa supply chain, focusing on Côte d'Ivoire – the current major producing country.

In Ghana, the governmental institution called Ghana Cocoa Board (COCOBOD) has a monopoly on exports and supply of processing factories; all cocoa beans must go through takeover points, i.e. COCOBOD warehouses located in major cities. Cocoa collection and transport to takeover points are managed by about 40 licensed buying companies (LBCs), each having purchasing clerks in cocoa communities (Nitidae and EFI, 2021). Despite this strongly monitored system, the level

of public disclosures by companies or public actors in Ghana has however proven insufficient to produce publicly-available tracing of cocoa back to its district of origin (Guye et al., 2023).

The traceability landscape evolves rapidly, as demonstrated by the development of the state-led national traceability systems aimed at tracing cocoa up to farmers, both in Côte d'Ivoire and Ghana – though companies show limited engagement with these systems which are yet to be fully comprehensive and sufficiently transparent to be deemed reliable to show compliance to the EUDR (EFI, 2024).

In **chapter 2**, company- and state-led cocoa traceability systems, often presented as solutions to deforestation, are discussed, highlighting several loopholes which may undermine their effectiveness in curbing deforestation.

1.3.1.5. Insufficient effectiveness in curbing deforestation

The central role of FSPs in sustainability approaches raises the question of their effectiveness. In practice, supply chain-focused policies have had only limited success regarding environmental outcomes (Garrett et al 2018). Their effectiveness hinges on multiple elements (Garrett et al 2019).

First, the **commitment characteristics** are crucial; (i) the *scope*, i.e. a large combined market share of committed firms (in number of forest-risk commodities, and volumes of each commodity), a large geographical reach (including all biomes at risk of deforestation), covering direct and indirect suppliers; (ii) the *target stringency*, i.e. zero-gross deforestation target with an immediate cut-off date; (ii) the *monitoring stringency*, i.e. relying on a functional, transparent system of compliance monitoring and verification (Garrett et al., 2019). Trade-offs exist between the scope and stringency of FSPs, with more stringent FSPs risking lower adoption by supply chain actors (Borck & Coglianese, 2009). This highlights the intrinsic limitation of voluntary sustainability schemes.

Second, the **additionality** of the commitment is key (i.e. whether it provides additional conservation benefits beyond business as usual), which is most likely if the target region has high forest-to-agriculture conversion rates, and by overcoming selection bias (Garrett et al., 2019). Selection bias may indeed undermine FSPs additionality; farmers with less forest remaining and larger properties, those already connected to well-functioning cooperatives (Garrett et al., 2021), already compliant (Lambin et al., 2018) or sufficiently wealthy to bear the costs of adoption and compliance (McDermott, 2013), can more easily adopt necessary practices to continue supplying to companies with FSPs, making them more likely and able to participate in conservation initiatives (Winters et al., 2015). This can also

increase local inequalities, with the poorest and most vulnerable community members least likely to have access to high quality farmlands, and to receive government extension and subsidized inputs (Hirons et al., 2018).

Third, the **political and socio-economic context** plays a central role in how FSPs eventually perform, as evidenced by election years driving deforestation rates up (Morpurgo et al., 2023). Co-occurring public policies may incentivize or impede deforestation (Garrett et al., 2021; Sandker et al., 2012). Many interventions rely on funding and political will, which may seize at any time, with undesired behaviours likely to revert (Garrett et al., 2022). This lack of permanence prevents effectiveness.

Several other barriers may also impede FSPs effectiveness, such as the lack of **adoption** and **compliance** by targeted actors, or **spillovers** which may offset the net conservation benefits of FSPs (cf. section 1.4).

Ultimately, most studies report only modest or no on-farm deforestation reductions (Garrett et al., 2021; Meemken et al., 2021). Some studies assessing beyond-farm deforestation dynamics, at landscape- or national-level, show mixed results depending on the context, time-frame, FSP considered, or on how spillovers are accounted for¹⁸. Critics suggest that the lack of effectiveness can be attributed to the fact that private FSPs promote business interests ahead of sustainability (van der Ven et al., 2018). Further, others suggest that private governance approaches constrain public governance (Cashore et al., 2021). For example, sectors with a high penetration of private governance initiatives can resist regulatory efforts with the argument that existing approaches already tackle the sustainability challenges faced by the sector (Bager, 2022).

¹⁸ A study looking at the impacts of seven FSPs on primary forest loss in tropical countries found lower loss associated to Fairtrade certification and larger loss for UTZ, RSPO, and Organic (Dröge et al., 2024). In Indonesia, Rainforest Alliance (RA) certified plantations and their surroundings had no significant differences in tree cover loss, canopy height, or vegetation indices as compared to non-certified ones, and no significant trends in vegetation indices were observed following certification (Dröge et al., 2025). Another study on coffee in Ethiopia however found that RA certification resulted in reduced degradation in nearby forests (Takahashi and Todo 2017). In Brazil, studies give heterogenous results for the Soy Moratorium; e.g., limited deforestation spillovers within and across nearby soy and cattle properties in Mato Grosso (Gollnow et al., 2018), domestic leakages offsetting about half of the avoided deforestation induced by the Moratorium (Villoria et al., 2022), no change in the exposure of companies or countries adopting ZDCs to soy-associated deforestation in the Cerrado (zu Ermgassen et al., 2019), cattle pasture deforestation being driven by the expansion of soy, as well as sugarcane and tree plantations at the country-level (zu Ermgassen et al., 2024). The RSPO in Indonesia was found to reduce deforestation within companies' certified supply bases as well as in their non-certified forest estates properties (where deforestation is forbidden), but increased deforestation in adjacent agricultural lands – resulting in, overall, an insignificant net total direct and indirect impact of RSPO certification on deforestation (Heilmayr et al., 2020).

In the cocoa sector in Western Africa, most studies look at income and agroforestry outcomes, with little evidence on the effectiveness of FSPs on forest conservation, due to low monitoring and disclosure by cocoa companies on the deployment of FSP activities (cf. section 1.3.1.4) (Addoah et al., 2025). On agroforestry, UTZ, RA and Organic certifications were found to have no impact on adoption (Thompson et al., 2022). The survival of distributed trees is likely very low (Kouassi et al., 2023; Ruf et al., 2021; Sanial et al., 2020). Despite considerable efforts by companies being reported (WCF, 2023), *cocoa monotypic landscapes* (Ajagun et al., 2022) are not transitioning to agroforestry (Becker et al., 2024) – which can be explained by a large range of factors, from institutional barriers to lack of farmer inclusiveness in policy design (Kouassi et al., 2023; Ruf, 2021; Sanial, 2018).

Chapter 3 takes a step back from individual company supply chains and looks at the entire cocoa region of Ghana, with the aim of understanding whether a sector-wide ZDC, covering all cocoa suppliers across the landscape, would be able to significantly reduce deforestation. Section 1.4 further details the considered spill-over mechanisms which are the focus of that chapter.

Moving to the global scale, next section introduces interventions considered within the global sustainability agenda, and their main implications for land and people, which are the focus of **chapter 4**.

1.3.2. The global sustainability agenda

1.3.2.1. Overview of interventions

Governments and private companies are increasingly making pledges aiming at mitigating the ongoing ecological and climate crisis, under the umbrella of three major interdependent United Nations conventions' targets: the Convention on Biological Diversity (CBD)'s Kunming-Montreal Global Biodiversity Framework (GBF) aiming at restoring at least 30% of degraded ecosystems and at conserving at least 30% of terrestrial and marine areas by 2030 (CBD, 2022); the UN Convention to Combat Desertification (UNCCD) land degradation neutrality (LDN) aiming for no net loss of healthy and productive land by 2030 (UNCCD, 2015); and the UN Framework Convention on Climate Change (UNFCCC) Paris Agreement to hold global warming well-below 2°C and pursue efforts to limit it to 1.5°C (UNFCCC, 2015). Three categories of land-based sustainability interventions are expected to be central in the pursuit of land, biodiversity and climate targets, and are shortly described hereafter.

(i) The conservation of natural ecosystems

Conservation is a major tool contributing to halting biodiversity loss and climate change (IPBES, 2019). An important turning point was the approval of the GBF Target 3, so-called ‘30x30’ target, aiming to conserve 30% of the world’s terrestrial and marine areas by 2030 (CBD, 2022). In addition, the UNCCD’s Glasgow Declaration (UNCCD, 2021) and UNFCCC’s first Global Stocktake (UNFCCC, 2024) committed to working towards halting and reversing deforestation and forest and land degradation by 2030. This highlights how healthy land and ecosystems are central to these three UN Conventions. Calls for the expansion of the global conservation target to cover half of Earth have also gained traction (Wilson, 2016). Target 3 aims for protected areas which ‘*must be managed with the primary objective of achieving positive outcomes for biodiversity*’ (CBD, 2022). Yet, conservation can translate into a wide range of realities, from strict nature reserves (IUCN category Ia) to protected areas with sustainable use of natural resources (IUCN category VI).

(ii) The restoration of degraded ecosystems, and afforestation focused on carbon capture

The restoration of degraded terrestrial ecosystems is increasingly recognized as a key strategy to address climate change, land degradation, biodiversity decline, and associated ecological and social challenges (IPBES, 2019; IPCC, 2023a).

Following a broadly accepted definition, ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (SER, 2004). However, ‘restoration’ captures a large range of understandings and realities. Human motivations and behaviours surrounding restoration are *enculturated* and *enearthed* – i.e., influenced by their cultural and social context (Löfqvist et al., 2023). Different actors therefore value different outcomes of restoration. This translates into a continuum of approaches, having more or less conflicting objectives, with, at one end of the spectrum, carbon stocks restoration (e.g., fast-growing tree plantations for carbon farming) and, at the other end, nature restoration that are primarily nature and biodiversity focused and may or may not have a positive carbon impact. In the intermediate space, nature-based solutions focused on restoring nature’s contributions to people are, in principle, restoration activities that can lead to both carbon capture and biodiversity benefits. Afforestation refers to planting trees on land which has not been forested in recent history (usually at least 50 years), while reforestation refers to the replanting of trees on more recently deforested land (UNFCCC, 2009).

There is a growing momentum driving the expansion of land use for ecosystem restoration and afforestation. In addition to the GBF Target 2 aiming at restoring at least 30% of degraded ecosystems by 2030, the UNCCD and SDG 15¹⁹ targets on land degradation neutrality, 2021–2030 was declared the Decade on Ecosystem Restoration. Besides, a myriad of initiatives have emerged such as the Bonn Challenge, which aims to restore 350 million hectares of land by 2030, the Great Green Wall in the Sahel region, the African Forest Landscape Restoration Initiative, AFR100 or the Trillion Trees Campaign (van der Esch, 2022).

Carbon dioxide removal (CDR) approaches, or negative emission technologies (NETs), have gained popularity among decision-makers and academics as measures to complement emission reductions and achieve net-zero emissions.

CDR is defined as a ‘*human activity capturing carbon dioxide (CO₂) from the atmosphere and durably storing it in geological, terrestrial, or ocean reservoirs, or in products*’ (Vaughan et al., 2024). CDR activities are made of (i) conventional approaches, i.e., methods that are well established, such as afforestation, reforestation, restoration of peatlands and wetlands and soil carbon sequestration, and (ii) novel approaches, which are generally at an earlier stage of development, including bioenergy with carbon capture and storage (BECCS), biochar, or Direct Air Carbon Capture and Storage (DACCS) (Vaughan et al., 2024). Most countries currently deploy or include plans to deploy conventional CDR – currently accounting for ~99.9% of all CDR carbon removals, with the vast majority being afforestation/reforestation (Vaughan et al., 2024).

(iii) The development of renewable energy infrastructures, with their related mining footprint, and the expansion of land-based bioenergy

Most climate stabilisation scenarios rely on a steep rise in global renewable electricity production by 2050, mainly based on solar and wind power (IEA, 2024). The International Energy Agency forecasts that the rise of the global electricity demand by 2030 will be equivalent to adding the current level of demand in the United States and the European Union (IEA, 2022). These technologies have substantially lower energy densities than traditional fossil fuels (Fritsche et al., 2017; van Zalk & Behrens, 2018) and their production is substantially more material-intensive (Kleijn et al., 2011) – meaning that future production will also escalate demand for many minerals (Kleijn et al., 2011; Luderer et al., 2019). It is unlikely that these new demands will be met by diverting use from other sectors or from

¹⁹ ‘Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss.’

recycling materials alone (Ali et al., 2017; Reck & Graedel, 2012). Consequently, there are growing concerns about the impact of the spread of renewable energy infrastructure on the global land system (Sonter et al., 2020; Stevens et al., 2017).

Besides, bioenergy is included in many IPCC scenarios to stay within the Paris Agreement, serving as fossil fuel substitute and, when combined with Carbon Capture and Storage (as BECCS), as a way to absorb carbon (IPCC, 2022b). Despite demonstrated limitations of its effectiveness in mitigating climate change and potential adverse effects including on water and food prices (Humpenöder et al., 2018), these scenarios dedicate vast tracks of land to energy crops (Creutzig et al., 2021).

1.3.2.2. Implications on land and people

The millions of hectares of land pledged by governments might provide a number of benefits for human societies, but also entail many trade-offs with existing land uses and basic human needs, such as food production or water regulation, as outlined in the first section of this introductory chapter. These expanding ‘green’ land uses may be considered themselves as *frontier* processes, playing out on the ground as an additional land demand exercising pressure on other land uses and actors (Buchadas et al., 2022). The drive to meet these targets often fails to fully account for social-ecological contexts, including challenges related to governance or equity issues, among others; e.g., for restoration (Osborne et al., 2021), conservation (Schleicher et al., 2019), or mining projects for renewable energy technologies (Sovacool et al., 2020). This can lead to land dispossession (Igoe & Brockington, 2007; McElwee, 2009; Woods, 2019), and reinforce existing inequalities (Fisher et al., 2018).

Though these issues are not new, the rising global sustainability agenda’s momentum intensifies *green grabs* (IPES-Food, 2024); i.e. the appropriation of land by governments and large companies to implement these interventions – thus excluding local land users and small-scale food producers. Like land grabs, green grabs lead to ‘*the restructuring of ownership, rights, and control of land and resources that were either privately, publicly, or commonly owned – or not owned at all – into the hands of more powerful actors*’ (IPES-Food, 2024). Land transfers for biofuels, renewable energy technologies (e.g. solar parks), and conservation schemes are now the dominant type of large-scale land acquisition, accounting for ~20% of the *recorded* land deals – outstripping conventional land grabs (IPES-Food, 2024).

Moreover, global sustainability interventions may be creating new forms of ecologically unequal exchange. Many global initiatives are promoted by the Global North, with large shares of commitments outsourced to locations in the Global South; e.g., conservation spending has increased substantially recently, with major funding flowing from the Global North to the Global South (Kuemmerle et al., 2019), and many emission offset pledges by countries and firms rely on land in other countries (Climate Resource, 2023).

These issues led to rising calls for the use of an *equity lens* in implementing these ‘green’ interventions, putting the participation, engagement and well-being of local people at the centre (Löfqvist et al., 2023). Calls for *connected conservation* also highlight the need to consider local-level conditions, but further stress the need to address distant drivers of biodiversity loss, such as international land demand, in conservation actions (Carmenta et al., 2023).

Chapter 4 assesses the expanding land uses for interventions deployed under the global sustainability agenda by 2050, and quantifies overlaps with other land uses, but also with each other, thus providing insights on potential additional trade-offs and conflicts which may arise. The next section delves into the complexities of land systems and how the expansion of land uses, whether for commodities such as cocoa or for sustainability interventions, can lead to multiple forms of spillovers.

1.4. Spillovers in complex land systems

Growing demands for land and the globalized displacement of land uses can initiate cascading effects. This section goes further into the intrinsically complex nature of land systems, providing a background on spillover processes involving distant causes and indirect consequences.

In land system science, *spillover* is an overarching term used to describe situations where ‘*land-use changes or direct interventions on land use (e.g., policy, program, new technologies) in one place have impacts on land use in another place*’ (Meyfroidt et al., 2018, 2020). Various forms of spillovers exist, including indirect land-use change, leakage and rebound effects. Land use displacement is also a form of spillover, which as a broad term can refer to describe these distant impacts when there is no demonstrated or existing causal links with a given intervention or initial land-use change (Meyfroidt et al., 2020).

Below, I focus on negative spillovers – i.e. spillovers that reduce the intended effect of the intervention, referred to as leakage –, though positive spillovers are also possible and arise when policies lead, for instance, to increased trust and learning

among people, beyond the population targeted by the policy, resulting in enhanced conservation and livelihood benefits (Garbach et al., 2012).

1.4.1. Indirect land use changes

Indirect land use change (iLUC) is a form of spillover where land use change in one place is caused by land use change in another place (Lapola et al., 2010; Meyfroidt et al., 2020; Searchinger et al., 2008). These complex land use dynamics depend on the competitiveness of different crops and land uses, land suitability, opportunity costs, and policies (Meyfroidt et al., 2018; Turner et al., 2020).

Beyond driving deforestation directly by expanding into forests, land uses can cause deforestation indirectly by competing and expanding into other land uses that are themselves displaced into forests (Kuschnig et al., 2021; Meyfroidt et al., 2013, 2018). This is the focus of **chapter 3**, which dives into land use dynamics in the tropical moist forests of Ghana, where cocoa expansion is taking up a considerable share of available farmland and is linked to food crops being displaced in neighbouring forests, raising concerns about food availability for households converting their food crops into cocoa plantations (Ajagun et al., 2022; Ashiagbor et al., 2025; Kumeh et al., 2022). Indirect land use change is also discussed in **chapter 4**, as they can be induced by large-scale land acquisitions linked to the unfolding of the global sustainability agenda.

1.4.2. Leakage

Leakage is a form of negative spillover, which arises when a policy implemented in a region increases incentives to do what the policy prohibits in non-targeted regions, resulting in a shift of environmental and social impacts across actors and space (Meyfroidt et al., 2020).

A land use leakage occurs through indirect land use change; it is ‘*caused by a land-use intervention, such as an environmental conservation policy, which triggers land-use change elsewhere that reduces the overall benefit of the local intervention*’ (Meyfroidt et al., 2018). Leakage is a challenge for policies aiming at reducing deforestation, such as protected areas or zero-deforestation commitments.

Four leakage mechanisms, potentially interacting, have been identified (see Meyfroidt et al., 2018):

(i) **Activity leakage** occurs when labour and capital used on the land targeted by the restrictions are reallocated to places with available and accessible land (Pfaff

& Robalino, 2017). The ‘pollution haven hypothesis’ is a well-known example; in reaction to environmental regulations, polluting companies move their activities to places with less restrictions (le Polain de Waroux et al., 2016). In agriculture, this leakage can be facilitated by growing market demand and unfavourable conditions for intensification – thus incentivizing farmers to continue production elsewhere.

(ii) Similarly, when the policy triggers a price increase and that productivity stagnates in producing regions, land use can expand elsewhere (assumed to be different actors) through a **commodity-market leakage** – depending on the degree of market integration and the price transmission for this good. Such leakage may also occur if the affected good is substituted by more land demanding goods.

The activity and commodity-market mechanisms can also happen under indirect land use change, when a first land use is replaced by a second land use as a result of a policy supporting the good produced by the second land use, or any other process increasing the demand for that second good; e.g., biofuels (Lapola et al., 2010).

(iii) Similar to commodity-market leakage, the **land-market leakage** happens when there is a market transmission of the increased land rent from the region(s) affected by the policy to land situated elsewhere, driving land investments, including deforestation, in these places (Miranda et al., 2019; Richards, 2015).

(iv) Finally, a **supply chain leakage** occurs when a supply chain policy meant to curb specific practices (e.g. pesticide use, deforestation) or shift away from a given good does not refrain producers to continue producing the same good or using the same practices but see them rather shifting to other buyers (also referred to as *market segmentation*) (Villoria & Hertel, 2011), selling their products by laundering them (Rausch & Gibbs, 2016), or pursuing even more unsustainable land use activities as a result of exclusion (Garrett et al., 2021).

1.4.3. Rebound effects

A **rebound effect** is a form of spillover where adoption of intensifying practices stimulates land use expansion (Angelsen et al., 2001). Though not the focus of this thesis, rebound effects play a pivotal role in policy effectiveness. In the cocoa sector, intensifying production is often seen as key to reducing deforestation (Addoah et al., 2025). Intensification is meant to reduce pressure on land through the so-called *Borlaug hypothesis of land sparing*; i.e. increasing productivity enables a reduction of the land area required to fulfil a given demand of land-based products, thus releasing land for other uses including nature preservation (Meyfroidt et al.,

2018). On the other hand, the *Jevon's paradox* or *rebound effect* (Alcott, 2005) occurs when ‘an increase in the productivity of one factor leads to its increased utilization’ (Rodríguez García et al., 2020); e.g. the intensification of cocoa production leads the increased productivity of cocoa farms, thus potentially increasing its profitability (given that the productivity increase does not come at too high costs), in turn propelling the expansion of cocoa farms.

1.5. Objectives and outline of the thesis

This doctoral research seeks to further elucidate the complexities linked to specific sustainability initiatives and policies, and question their potential as ‘solutions’ to the ecological and climate crisis, in particular by exploring their implications in terms of increasing pressure on land, in places sometimes far from where decisions or consumption happen.

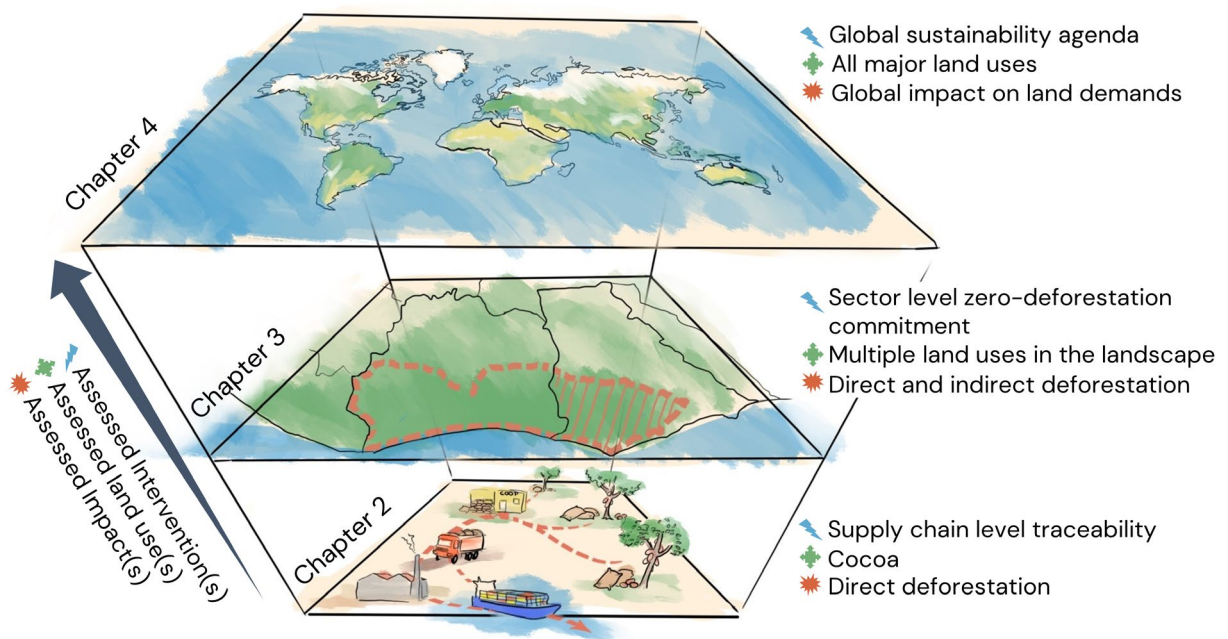


Figure 1.2. Schematic summary of the scope of chapters 2, 3 and 4 broadening incrementally in terms of geography, interventions, land uses and impacts assessed (drawing by Michaël Dubois).

As introduced in section 1.1, each chapter broadens the geographic scale and the assessed interventions, land uses, and impacts (Figure 1.2), i.e., (i) first, at the individual supply chain level, with company-led traceability, assessing cocoa and its direct deforestation impact (**chapter 2**); (ii) second, at the sector level, with zero-deforestation commitments, assessing cocoa and other major land uses and their direct and indirect deforestation impact (**chapter 3**); (iii) third, at the global level,

with major land-based interventions considered within the global sustainability agenda (e.g. conservation, restoration, renewable energy), assessing their impact on land demand (**chapter 4**). **Chapter 5** summarizes the findings and explores underlying causes of the climate and ecological breakdown and avenues for systemic change. Further details for each chapter are given below.

The first two core chapters focus on the cocoa sector and its associated direct and indirect deforestation in Côte d'Ivoire and Ghana, the two major producing countries (cf. section 1.2.2). These forest landscapes of West Africa typify the multidimensional sustainability challenge of halting deforestation while improving food security and livelihood opportunities.

Chapter 2 assesses the level of transparency and traceability in the Ivorian cocoa sector (cf. section 1.3.1.4), and discusses the potential of a supply chain level 'solution' – traceability – in ensuring deforestation-free sourcing.

This work was part of the Transparency for Sustainable Economies (Trase) initiative (www.trase.earth). We used publicly-available remote-sensing and supply chain data for Côte d'Ivoire, the world's largest cocoa producer, to quantify cocoa-driven deforestation and trace 2019 cocoa exports and the associated deforestation from their department of origin, via trading companies, to international markets.

Chapter 3 estimates the extent of displacement of food crops into forests due to cocoa expansion in Ghana (cf. section 1.4.1) and discusses the potential of a broader sector-level 'solution' – zero-deforestation sourcing – to stop deforestation in a context of limited land (cf. section 1.3.1.3).

We used satellite-based maps and secondary data to quantify the direct deforestation and forest degradation between 2000 and 2019 in the entire cocoa-growing region of Ghana which we attributed to the land uses detected in 2019. Then, we used a land-balance approach to assess the indirect role of the expansion of land uses, including cocoa, in deforestation and degradation for food crops.

These two first papers discuss the risks of leakages (cf. section 1.4.2) of deforestation-free sourcing commitments in the cocoa sector, with farm-level traceability being a major implementation tool. The results first highlight the complexity of traceability in the cocoa sector, which allows for negative spillovers, thus preventing deforestation from being halted, and then show the indirect role of cocoa in deforestation by occupying space, up to land saturation. Land being limited, it raises the question of how much cocoa can actually be produced sustainably, considering the multiple and growing human needs?

This question cannot be answered by focusing only on a specific region or country, as land uses may be displaced via global trade (cf. section 1.2). Nor can it be answered for an individual sector, as land uses may compete for the same land.

Chapter 4 thus enlarges the perspective and assesses at the global level major land demands linked to global sustainability targets, including conservation, tree planting and restoration, and renewable energy (cf. section 1.3.2) – and how they may compete with each other and with current land uses including cropland, grazing land and forestry, participating in the global land squeeze (cf. section 1.1) and increasing the risk of green grabs (cf. section 1.3.2.2). It discusses the limitations of these interventions as ‘solutions’ to the climate and biodiversity crisis.

We compiled quantitative targets of the considered interventions, expressed through multilateral agreements or public commitments. We spatialized these targets using global suitability or priority maps from recent works, selecting the best-suited locations for each target, per country. We derived maps of the distribution of each intervention’s priority location in 2050. By stacking these, we identified areas of overlap between considered interventions, as well as with major existing land uses.

After summarizing the main findings and limitations of the thesis, and reviewing the major issues of current conventional approaches to sustainability, **chapter 5** looks ahead; in the face of the globalized land squeeze, and based on what was learned in this research journey, I reflect on a broader question regarding land-based sustainability issues: ‘What can we do about it?’

This chapter seeks to further broaden the perspective by digging deeper in search for roots, or underlying causes, of the global land squeeze and, more generally, of the climate and ecological turmoil. It first further stresses how tackling sustainability issues requires a systemic approach, rather than focusing on each problem individually. It then points at how reducing production and consumption of high-income countries would be a very – if not the most – effective way of lifting pressures on nature, and argues that organising it should be a priority for human societies in coming decades. Several alternative economic perspectives and frameworks seek to achieve this reduction of energetic and material throughput – including degrowth, which is briefly explored. This chapter calls not only for out-of-the-box thinking, but also for building *societies of care*, reinforcing or recreating relationships among people and between people and the land.

Chapter 2

Transparency, traceability and deforestation in the Ivorian cocoa supply chain

This chapter has been published:

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Abstract

Cocoa production has been identified as a major global driver of deforestation, but its precise contribution to deforestation dynamics in West Africa remains unclear. It is also unknown to what degree companies and international markets are able to trace their cocoa imports, and satisfy their sustainable sourcing commitments. Here, we use publicly-available remote-sensing and supply chain data for Côte d'Ivoire, the world's largest cocoa producer, to quantify cocoa-driven deforestation and trace 2019 cocoa exports and the associated deforestation from their department of origin, via trading companies, to international markets. We find 2.4 Mha of cocoa deforestation and degradation over 2000-2019, i.e., 125,000 ha/y, representing 45% of the total deforestation and forest degradation over that period. Only 43.6% (95% CI: 42.6 - 44.7%) of exports can be traced back to a specific cooperative and department. The majority of cocoa (over 55%) thus remains untraced, either indirectly sourced from local intermediaries by major traders (23.9%, 95% CI: 22.9 - 24.9%), or exported by untransparent traders – who disclose no information about their suppliers (32.4%). Traceability to farm lags further behind, and is insufficient to meet the EU due-diligence legislation's proposed requirement for geolocation of product origins. We estimate that trading companies in the Cocoa and Forests Initiative have mapped 40% of the total farms supplying them, representing only 22% of all Ivorian cocoa exports in 2019. We identify 838,000 hectares of deforestation over 2000-2015 associated with 2019 EU imports, 56% of this arising through untraced sourcing. We discuss issues of company- and state-led traceability systems, often presented as solutions to deforestation, and stress the need for transparency and for the sector to work beyond individual supply chains, at landscape-level, calling for collaboration, stronger regulatory policies, and investments to preserve the remaining stretches of forests in West Africa.

2.1. Introduction

International trade of agricultural products drives land use change and deforestation in distant regions across the planet, with large sustainability impacts including carbon emissions and biodiversity loss (Pendrill *et al* 2019). Supply chains are often opaque, little is known about the origin of the products, and even less about the sustainability risks associated with their production (Gardner *et al* 2019).

Global sales of chocolate-based confectionery products were estimated at 110 billion USD in 2019, up 33% from 2008/09 (BASIC and FAO 2020). Côte d'Ivoire is the world's largest producer of cocoa, with about one million smallholder farmers producing annually over 2 M tonnes of cocoa beans – 40% of the world's cocoa harvest (FAO 2020). This trade is worth more than \$4.7 billion in freight-on-board value per year, over one-third of Côte d'Ivoire's entire export revenue (OEC 2020).

Yet, the crucial importance of cocoa in the Ivorian economy comes at a cost: Côte d'Ivoire lost 80% of its forest cover over the past 60 years (IFFN 2021). Cocoa has been flagged as a major commodity driving deforestation (Goldman *et al* 2020b, Pendrill *et al* 2022), but its precise contribution to deforestation in Côte d'Ivoire remains unclear. Pan-tropical studies provide coarse estimates of cocoa-driven deforestation, attributing tree cover loss to cocoa using national-level agricultural statistics (Pendrill *et al* 2019) or coarse land use data (Goldman *et al* 2020b). Within Côte d'Ivoire, studies are limited to specific regions (Andrieu *et al* 2018, Barima *et al* 2016, Brou *et al* 2005, Carin *et al* 2019, Kouassi *et al* 2021, Ongolo *et al* 2018, Ouattara *et al* 2021) or do not disentangle the specific role of cocoa in national deforestation (BNETD *et al* 2017, SEP-REDD+ CI 2017).

In response to civil society pressure, chocolate companies have made a series of zero deforestation commitments and investments in traceability, seeking to identify the origin of the cocoa they buy. The Cocoa and Forest Initiative (CFI), in particular, is a public-private partnership launched in 2017 by a consortium of 35 chocolate and cocoa companies together with the governments of Ghana and Côte d'Ivoire, aiming to contribute to ending deforestation and forest degradation in the cocoa supply chain (CFI 2021). The urgency behind efforts to trace cocoa 'from bean to bar' has been further increased by the European Commission's proposed due-diligence legislation, requiring companies importing deforestation-risk products (such as cocoa or chocolate) to demonstrate that imports do not originate from recently deforested nor degraded land (EC 2021).

The information that companies have (i.e. their supply chain ‘traceability’) and disclose about their cocoa supply chains (i.e. their supply chain ‘transparency’) is, however, disparate and it is challenging to assess where companies source their cocoa from, let alone monitor progress in implementing sustainable sourcing initiatives (Dontenville *et al* 2022). Here, as part of the Transparency for Sustainable Economies (Trase) initiative (www.trase.earth), we address this gap, pulling together several publicly-available datasets for Côte d’Ivoire’s 2019 land use and cocoa exports, to link cocoa production, and its associated deforestation, to specific companies and markets.

Farmers sell their cocoa to cooperatives, or to middlemen – ‘pisteurs’ – who sell on to larger-scale local buyers – ‘acheteurs’, also called ‘traitants’ – many of whom work unlicensed. Traders can buy ‘directly’ from cooperatives and licensed buyers (‘acheteurs agréés’), or ‘indirectly’ from other local intermediaries, before exporting to various countries (Nitidae and EFI 2021). What the sector calls ‘direct sourcing’ is thus traceable to the first buyer, cooperatives in most cases, but not necessarily to farm. To map the supply chain, we make a distinction between three sourcing types: (i) the cocoa that is traced to known farmer cooperatives or licensed buyers, which were disclosed by traders (called ‘direct’ sourcing by actors in the cocoa sector), (ii) cocoa which is indirectly sourced by those trading companies from other intermediaries, and is thus nearly impossible to trace under current traceability systems, and (iii) cocoa which is ‘unknown’, i.e. exported by ‘untransparent’ traders, who disclose no information about their supply chain (Figure 2.1).

Specifically, we tackle three research questions:

1. What is the state of traceability and transparency in the Ivorian cocoa supply chain?
2. What is the role of cocoa in recent deforestation in Côte d’Ivoire?
3. How is deforestation exposure distributed among actors (i.e. traders and importing countries)?

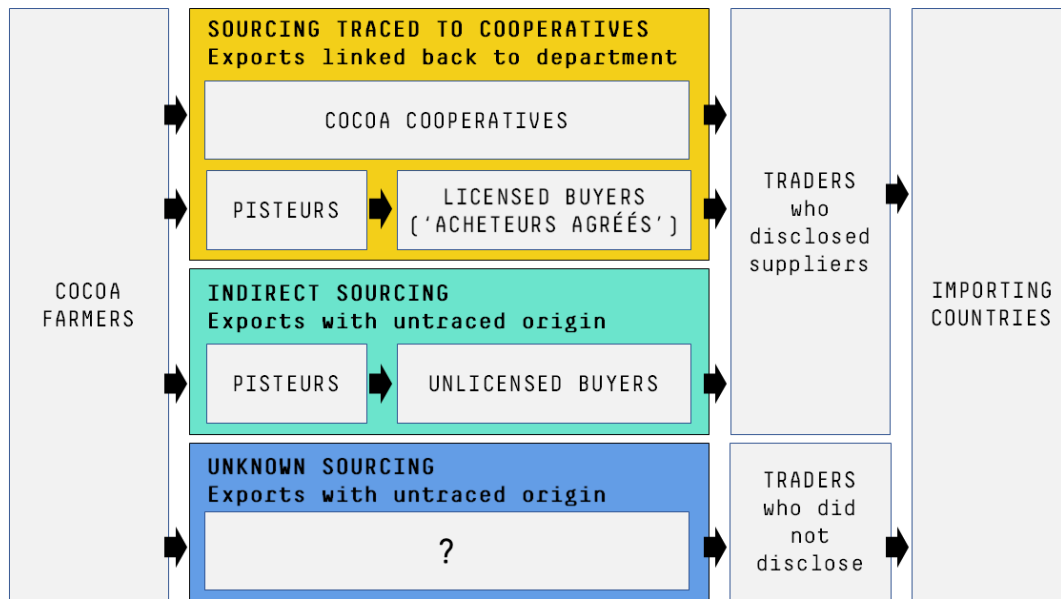


Figure 2.1. Schematic of the cocoa supply chain in Côte d'Ivoire with *Sourcing traced to cooperatives* in yellow, i.e. exports linked back to disclosed cooperatives, also including the small volumes sourced from known licensed buyers; *indirect sourcing* in green, i.e. exports that could not be linked back to cooperatives or buyers (purchased through unlicensed buyers) for traders who disclosed their suppliers; *unknown sourcing* in blue, i.e. exports of untransparent traders who did not disclose their suppliers.

2.2. Methods

We used the Spatially Explicit Information on Production to Consumption Systems (SEI-PCS) approach to link exports of agricultural commodities back to the jurisdiction of production (Godar *et al* 2016). We followed five steps (detailed in SI text S1): We first calculated total exports per trader and imports per importing countries. Second, we identified the intermediaries (cooperatives and ‘acheteurs agréés’) from which traders source from, and their location. Third, we estimated the volumes sourced from these intermediaries by the different traders, providing a map of the subnational origin of the cocoa traced to cooperatives. Fourth, we estimated the cocoa production per department, allowing to geographically distribute the cocoa indirectly sourced. Finally, we used remote sensing-based cocoa and deforestation maps to estimate deforestation due to cocoa (further: ‘cocoa deforestation’) across the country and link cocoa supplies with deforestation exposure.

2.2.1. Mapping cocoa sourcing

We calculated the volume of cocoa exported by distinct traders and imported by distinct countries using data on export shipments from 2019 (SI table S1). As no

information is publicly-available on the location of farms supplying each trader, we map the supply chain up to the first buyer, and not to the farm. For traders that are transparent about their ‘direct’ suppliers, we identified the list of cooperatives from which they source from, and the department in which they are located, using Mighty Earth’s ‘Cocoa Accountability Map’. This map includes information on 4451 cooperatives and lists, for 710 of them, companies that are known to source from them (SI Figure S2A, table S2). The latter comes from self-disclosures in 2019 and 2020 by major cocoa traders and processing companies. A smaller volume is purchased from ‘acheteurs agréés’; buyers licensed to source cocoa from specific departments (SI table S3). Their supply comes from an approximate known origin, much like a cooperative. When reporting the results, they are included under the term ‘**sourcing traced to cooperatives**’.

The volumes sourced by companies from individual cooperatives are estimated from the number of farmers per cooperative (provided for 592 cooperatives, SI Figure S2B), and production volumes per farmer. For the remaining 119 cooperatives with a known trading company but no number of farmers provided, we simulated this number using 1000 Monte Carlo estimates (random sampling with replacement) drawn from the set of disclosed cooperative sizes. Next, we used recent production data (kg of dry beans/farmer/year) collected from 441 farms across the country (Bymolt *et al* 2018) (SI Figure S3). We assigned a production volume to each cooperative-company connection using 1000 Monte Carlo estimates of this dataset, keeping the confidence interval. This calculation assumes that traders purchase the entirety of the cocoa of each of their supplier farmers (rather than each farmer selling to multiple traders), which provides an upper-bound estimate of the volumes traced to cooperatives. The latter are assigned to the department where the cooperative is located – thus assuming that the cocoa sourced by a distinct cooperative mainly comes from the department where it is registered.

We then estimated each trader’s proportion of **indirect sourcing** by subtracting the volume traced to cooperatives from the total exported volume. This provides an estimate of the level of traceability for those traders that disclosed their sourcing. For untransparent trading companies that did not disclose any information on their sourcing (43 companies of 73 in total), the total exported volumes are marked as ‘**unknown sourcing**’. This provides an estimate of the level of transparency in the sector.

As an additional analysis, we compiled the number of mapped farms reported in CFI signatories’ 2018-2020 progress reports and estimated the total number of

farms that these companies are sourcing from based on their traded volume in 2019 (SI text S1.5, table S4).

2.2.2. Quantifying the role of cocoa in recent deforestation

To calculate cocoa deforestation, we overlaid a remote-sensing cocoa land use map for the years 2019-2021 at 10 m resolution (Kalischek *et al* 2022) (hereafter, the ‘2019 cocoa map’) with a map of Tropical Moist Forest (TMF) produced by European Commission’s Joint Research Centre (JRC), depicting annually from 1990 to 2020 the extent of undisturbed tropical moist forest and its changes at 30 m resolution (Vancutsem *et al* 2021) (SI Figure S5).

We created a map of the land-use dynamics over the last 20 years including deforestation and degradation, their association with cocoa expansion, and the remaining undisturbed and degraded TMF cover. A degraded forest is defined by the JRC TMF dataset as a forest that has been temporarily disturbed (< 2.5 years), which can include regrowth, but excludes tree plantations (SI text S1.6.1). Here, ‘deforestation’ corresponds to pixels that switched from undisturbed TMF to another land cover between 2000 and 2019, and ‘degradation’ to pixels that switched from undisturbed to degraded TMF. From those, pixels detected as cocoa were classified as ‘cocoa deforestation’ and ‘cocoa degradation’, the latter also including cocoa pixels detected as undisturbed forest by the TMF map in 2019 (0.3 Mha). Cocoa degradation could be forest converted to cocoa but with a relatively dense remaining shade canopy cover, or shaded cocoa plantations that were too recently established to be classified as deforestation (requiring 2.5 years of detected disturbance).

We tested the sensitivity of our findings to different remote sensing products by repeating our analyses using two other published cocoa maps (Abu *et al* 2021, Vivid Economics 2019) and two other deforestation datasets (BNETD *et al* 2017, Hansen *et al* 2013). We used primarily the cocoa map and JRC-TMF maps because of their superior accuracy (SI table S5) and publicly-available methodologies.

2.2.3. Linking cocoa deforestation to sourcing

To link to the 2019 trade flows, we calculated the ‘cocoa deforestation’ as the area that lost its undisturbed TMF between 2000 and 2015 and is detected as cocoa in 2019. Thus, here, cocoa degradation and deforestation are merged into ‘cocoa deforestation’. Cocoa trees need three to five years to become fully productive, so

deforestation events after 2015 are not expected to have contributed to the 2019 cocoa harvest.

To link cocoa sourcing to cocoa deforestation, we calculated the relative cocoa deforestation per department, by dividing the annualised cocoa deforestation area by the 2019 cocoa production per department, to obtain annual hectares deforested per tonne of cocoa produced. In the absence of official publicly available data on subnational cocoa production in Côte d'Ivoire, we generated a map of cocoa production per department by weighting the 2019 cocoa map by a map representing the suitability to grow cocoa per pixel (Schroth et al. 2016) (SI Figure S6). We then allocated the national cocoa production (ICCO 2021) to each department according to their weighted cocoa area.

We calculated the 'cocoa deforestation exposure' of each trader or importing country, which provides a measure of the exposure of actors to cocoa deforestation based on their sourcing patterns. We multiplied the relative cocoa deforestation by the volume traced to cooperatives of each actor, per department, to obtain the cocoa deforestation exposure linked to their traced sourcing. We also calculated actors' annual relative cocoa deforestation exposure (i.e. number of hectares deforested per tonne of cocoa traced to cooperatives, divided by 15 years). The relative cocoa deforestation for untraced cocoa was estimated as an average of the relative cocoa deforestation per department, weighted by the proportion of total untraced volume supplied by each department. We multiplied this value by actors' untraced (unknown or indirectly-sourced) volumes to calculate each actor's exposure to cocoa deforestation linked to their untraced sourcing.

2.3. Results and discussion

2.3.1. What is the state of traceability and transparency in cocoa sourcing?

We trace 43.6% (95% CI: 42.6 - 44.7%) of Côte d'Ivoire's exports back to a specific cooperative and department, i.e. 875,468 tonnes (95% CI: 855,360 – 896,993 tonnes) of 2,006,406 tonnes exported in total in 2019. The majority of cocoa (over 55%) is thus untraced, either indirectly sourced from local intermediaries by major traders (23.9%, 95% CI: 22.9 - 24.9%), or exported by untransparent traders (32.4%), i.e. 'unknown' sourcing (Figure 2.2). Untraced sourcing dominates in the North-East and along the western border of the cocoa producing region of Côte d'Ivoire (80-100% untraced sourcing) (SI Figure S7).

In 2019, eight companies (Cargill, Barry Callebaut, Olam, Touton, Sucden, S3C, ECOM, and Africa Sourcing) handled 60% of exports (SI Figure S8-9). The share of indirect sourcing among traders varied between 25% (95% CI: 10.4 - 34.4%) (ECOM) and over 70% (95% CI: 66.4 - 75.1%) (Sucden), posing strong traceability challenges. There are also gaps in transparency. A third of the cacao was exported by untransparent traders, among which S3C and Africa Sourcing, as well as many of the smaller companies, who disclose no information about their suppliers. Even traders who did disclose their supplying cooperatives can greatly strengthen their level of transparency: their disclosed data is scattered, incomplete, irregularly or never updated, and non standardised. The Accountability Framework Initiative provides unified guidance to companies on reporting and disclosure practices to increase the credibility of their claims.

Looking at importing markets, European Union countries imported 60% of Ivorian cocoa beans in 2019, of which only 45.6% (95% CI: 44.5 - 46.8%) were traceable to cooperatives (Figure 2.3A-B). These data detail the country where cocoa was first imported – the Netherlands, for example, appears as the principal destination (27% of exports) because the port of Amsterdam is the main hub for cocoa imports into Europe, with cocoa then being re-exported or processed for consumption elsewhere. The rate of untraced imports among major importers varied between 36% (95% CI: 33.3 - 39.8%) (France) and 66% (95% CI: 66.2 - 67.1%) (Malaysia).

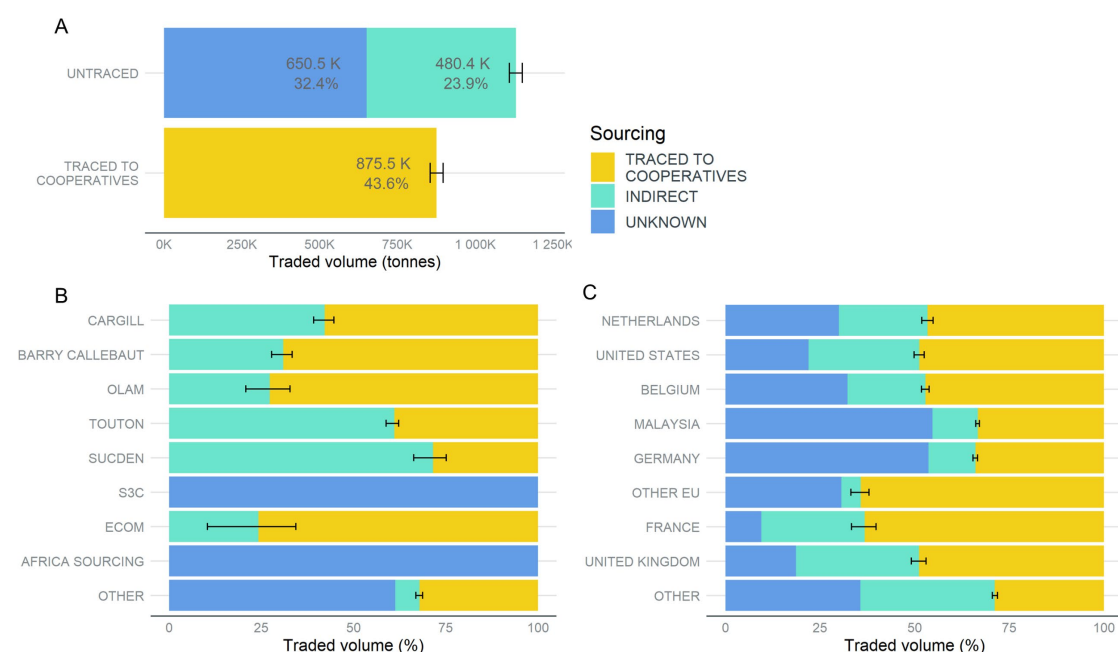


Figure 2.2. Share of 2019 cocoa sourcing traced to cooperatives, indirect and unknown sourcing. (A) total volume exported from Côte d'Ivoire in tonnes, (B) per major trading company and (C) per importing country in percentage. Error bars represent the 95% confidence interval of the estimates.

Traceability to farm lags further behind traceability to cooperatives. For the seven trading companies who disclose their farm-level mapping in Côte d'Ivoire as part of the CFI – handling 52% of the total traded volume – we estimated that 69% (95% CI: 64 - 74%) of the volumes they source ‘directly’ from cooperatives was mapped to farm-level (343,207 farms of the estimated 495,000 farms supplying them ‘directly’ through cooperatives (95% CI: 465K - 533K)). For comparison, this is consistent with the self-reported number across all CFI companies, which was 74% in 2020 and 72% in 2021 (CFI 2021). Yet, when including their indirect sourcing, this only represents 40% of the total 850,000 farms supplying these traders (95% CI: 845K - 854K), which in turn only represents 22% of all cocoa exports in 2019. Approximately 579,000 farms (95% CI: 575K - 583K) supply the EU market through these CFI traders (Figure 2.3C), compared to the 343,207 farms that they had georeferenced in 2020 (465,400 reported in 2021). This shows that current levels of farm-level traceability are insufficient to meet the EU’s future due-diligence legislation, expected to require geolocation of the product’s origin. Overall, 838,000 hectares of deforestation over 2000-2015 were associated with 2019 EU imports (Figure 2.3D). Nor is the EU the only market moving towards due-diligence for deforestation-free imports: similar legislation is also tabled in the United States, UK, Canada and Japan (OECD 2021) – the destination of 11.6, 4.6, 3.2 and 0.1% of Ivorian cocoa exports, respectively.

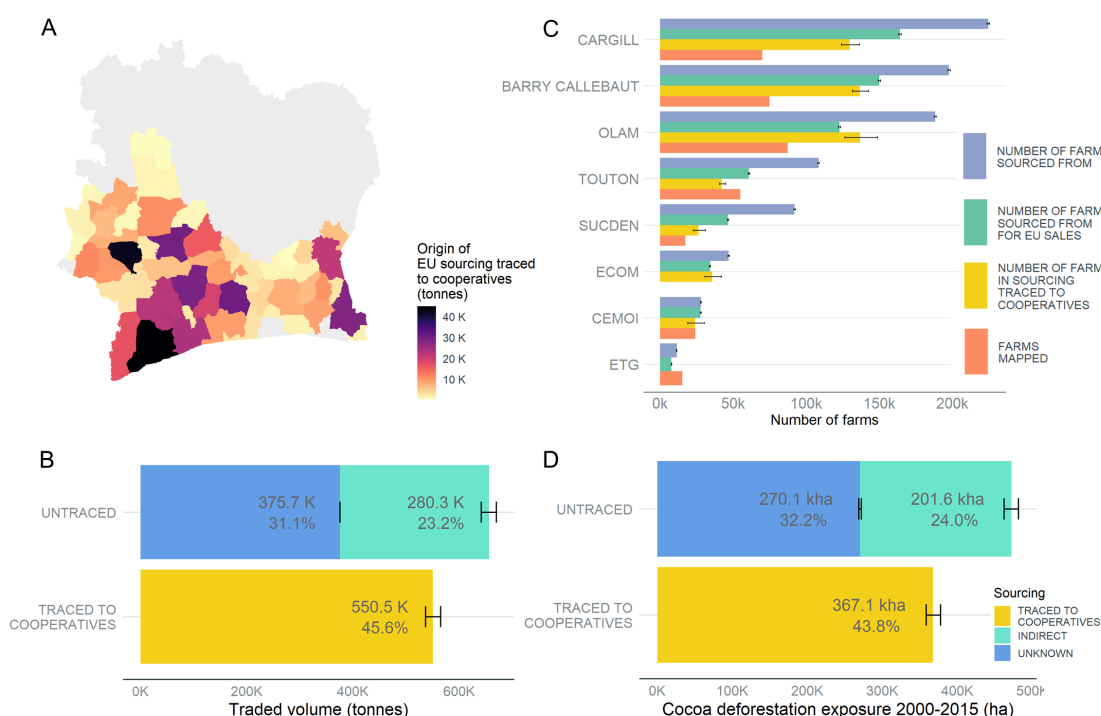


Figure 2.3. Data for the European Union market. (A) Map of the EU sourcing traced to cooperatives per department. (B) Volume imported into the EU in 2019 by type of sourcing. (C) Comparison of the number of farms mapped by CFI traders as reported in the 2018-2020 CFI reports (orange), the estimated total

number of farms they sourced from in 2019 (blue), the estimated number of farms they sourced from for the EU market (green), and the estimated number of farms in their sourcing traced to cooperatives (yellow). Note: ECOM does not disclose the number of farms mapped in Côte d'Ivoire. (D) Cocoa deforestation exposure in hectares embedded in the EU imports in 2019, per sourcing type. Error bars represent the 95% confidence interval of the estimates.

2.3.2. What is the role of cocoa in recent deforestation?

We found 2.4 Mha of cocoa deforestation and degradation over 2000-2019 (0.125 Mha per year), of which 1.25 Mha is cocoa degradation. This does not include the deforestation of already degraded forest in 2000. The remainder is linked to other agricultural crops such as palm oil, rubber or food crops, logging and mining (SEP-REDD+ CI 2017). Over that period, 5.2 Mha of the initial 6.7 Mha of undisturbed TMF in 2000 was lost (i.e. 77%), either degraded or deforested, leaving the country with less than 1.5 Mha of undisturbed tropical moist forest in 2019, representing 4.7 % of its territory. This percentage increases to 9.4% when accounting for the 1.5 Mha of degraded TMF (Figure 2.4A). The latest national flora and fauna survey gives a similar total natural forest area of 2.5 Mha in 2020 (without wooded savanna) (IFFN 2021).

Some limitations inherent to spatial datasets may affect the results. For instance, the removal of shade trees or the rehabilitation of old abandoned cocoa farms may be categorised as cocoa deforestation or degradation, because highly shaded farms in 1990 (the starting point of the JRC TMF time series) may have been classified as undisturbed TMF. On the other hand, young cocoa farms are, to some extent, not detected by the cocoa map – thus very recent deforestation for cocoa is certainly classified as non-cocoa deforestation or degradation.

Drivers of land use changes are complex and deforestation or forest degradation rarely have a single underlying cause (Pendrill et al 2022a). It is possible that other land uses occurred in the period between deforestation or degradation and the detection of cocoa in 2019 – such as other crops on deforested land, or timber extraction or the planting of highly shaded food crops on forests detected as degraded. We report, however, ‘cocoa deforestation’ and ‘cocoa degradation’, because cocoa was the major expanding land use during this time period, occurring on 45% of land deforested or forest degraded between 2000-2019, and other studies also find cocoa as the main driver of deforestation in Côte d'Ivoire (Barima et al 2020, SEP-REDD+ CI 2017, Higonnet et al 2017, Ruf and Varlet 2017). Cocoa was identified as the major export-oriented crop expanding in Sub-Saharan Africa in 2000–2013, at an annual rate of 132,000 ha, with Côte d'Ivoire being one of the countries most at risk of deforestation because of its low available cropland outside forest areas

(Ordway *et al* 2017). Pantropical studies estimated annual cocoa deforestation rates in Côte d'Ivoire of 33,000 ha/y (Goldman *et al.* 2020b) and 13,000 ha/y (Pendrill *et al* 2022) over 2001-2015, while a perception-based study estimated the role of cocoa in deforestation at 23% (the highest of all drivers) between 1990 and 2015 (Nitidae 2016). These are considerably lower than our 125,000 ha/y of cocoa deforestation and degradation over 2000-2019.

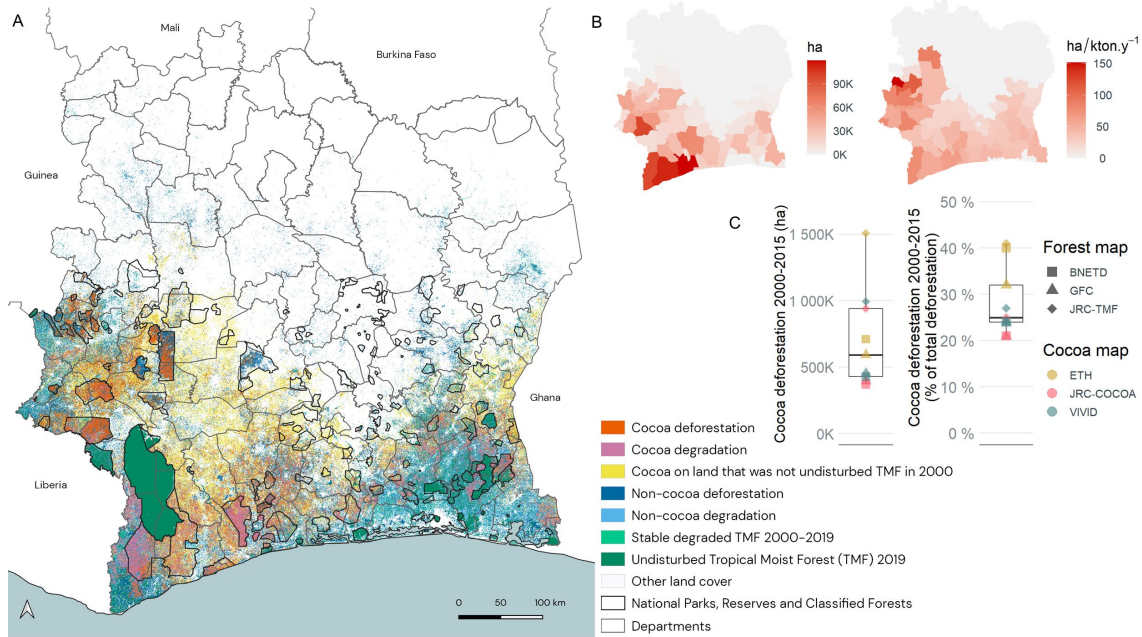


Figure 2.4. Deforestation maps. (A) Map of cocoa and non-cocoa deforestation and degradation between 2000 and 2019 in Côte d'Ivoire. 'Other land cover' includes forest already degraded in 2000 (data sources: EC JRC, Kalischek *et al.* 2022, Ministère des Eaux et Forêts for protected areas). (B) Cocoa deforestation (2000 to 2015) in hectares (left) and Annual relative cocoa deforestation in hectares per year per kilo tonne of cocoa produced by the department in 2019, per department with more than 500 tonnes of cocoa produced in 2019 (right). (C) Variations of results following the nine combinations of three cocoa maps and three forest maps, with the 2000-2015 cocoa deforestation (left) and the percentage of the total 2000-2015 deforestation being due to cocoa (right). [Higher resolution available in the online version]

Côte d'Ivoire has experienced several waves of cocoa expansion, first in the East, then expanding in the Southwest in the 1970s. The most recent belt of expansion in the Northwest, along the Guinea and Liberia borders (Sanial, 2017), has heavily impacted protected areas (Figure 2.4A) where the few remaining forests have been concentrated (BNETD *et al* 2017). This is also mirrored by the concentration of cocoa deforestation in the western departments, with Sassandra, San-Pédro, Tabou and Guiglo amounting for 28.5% of the total cocoa deforestation over 2000-2015 (Figure 2.4B). Over the 2000-2019 period, we found that protected areas (including classified forests) in the cocoa growing region lost in total over 60% of their undisturbed TMF of 2000 (representing 1.3 Mha), with 62% of this deforestation

and degradation being due to cocoa (SI table S5). In 2019, 60% of the remaining protected undisturbed TMF was sheltered by the Taï National Park – the best preserved protected area of the country. 25% of the cocoa area in Côte d'Ivoire was located within protected areas and the area under cocoa production has still been expanding (FAO 2020, Vivid Economics 2020). Accounting for higher yields on recently cleared land (Ruf 1987), cocoa in protected areas very likely represents more than 25% of the national production of the world's largest cocoa producing country.

Multiple factors can explain the high rate of cocoa deforestation and degradation, including reduction in soil fertility and yields, migration, the lack of law enforcement, corruption, the lack of financial means and the 2000's political instability (Andrieu *et al* 2018, Barima *et al* 2016, Ongolo *et al* 2018, Ruf *et al* 2015). However, beyond all these factors, cocoa deforestation is rooted in the increasing international demand for cocoa.

2.3.3. How is cocoa deforestation exposure distributed among actors?

To link cocoa deforestation, here corresponding to all areas of undisturbed TMF lost to cocoa and thus also including areas detected as cocoa degradation, to 2019 trade flows, we consider the 2000 to 2015 period, during which 55% of the 2000 undisturbed TMF area was lost (from 6.7 Mha to 3 Mha). This is substantially more than the loss of forest cover reported by the National REDD+ Secretariat, i.e. from 4.7 Mha to 2.7 Mha (BNETD *et al* 2017). This difference may be due to differing methodologies and forest definitions (see SI table S6). In total, 1.5 Mha were replaced by cocoa over that period (40.7% of the undisturbed TMF lost), at a rate of 100,000 ha per year. The relative cocoa deforestation (ha/tonne produced) is quite homogenous across the country, with slightly larger values in the West (Figure 2.4B).

57.7% of the total cocoa deforestation exposure is attributed to untraced sourcing (24.5% to indirect and 33.2% to unknown sourcing), which, in relative terms, represents a slightly higher risk of being exposed to deforestation than sourcing traced to cooperatives (47.9 compared to 45.4 ha/kton.y⁻¹) (Figure 2.5A). Cargill, Barry Callebaut, Olam and Touton each have 100-180,000 ha of cocoa deforestation exposure linked to their 2019 sourcing, but the exposure per tonne traced to cooperatives is similar for all traders (Figure 2.5B). The exception is Touton whose higher relative cocoa deforestation exposure is due to its sourcing pattern concentrated on few departments among which some faced high cocoa deforestation, e.g.

Guiglo – from which 18% of Touton’s sourcing traced to cooperatives came from. The eight CFI traders, handling 55% of the total exports, encompass 53.4% of the total cocoa deforestation exposure. Their sourcing traced to cooperatives has a slightly lower relative exposure than non-CFI signatories’ (44.3 compared to 48.6 ha/kton.y⁻¹ for non-CFI traders). As the CFI was signed in 2017, this difference should be considered as a baseline and not as the impact of the agreement (SI Figure S10).

Deforestation exposure trickles down to importing countries, with a large share reaching the Netherlands and its cocoa processing facilities (Figure 2.5C). In Germany, 72% of chocolate confectionery was certified according to sustainability standards in 2019 (79% in 2021) (BDSI 2021), but we show that the country imported 101,000 ha of 2000-2015 deforestation. Here, we mapped *physical* flows of cocoa imports, whereas certification credits for several certification standards are decoupled from physical flows, leaving room for deforestation to enter the market.

Tracing cocoa and linking sourcing to deforestation comes with some qualifications. We tested the sensitivity of our findings by repeating our analyses using different remote sensing products (SI text S2, table S6-7, Figure S11-16). The combination of the ETH²⁰ and JRC-TMF maps, which are the most accurate maps available for deforestation and cocoa plantations, shows a higher rate of cocoa deforestation than other datasets as ETH detects more cocoa and JRC-TMF more forest loss – notably because it includes forest degradation (Figure 2.4C). But, importantly, regardless of the magnitude of cocoa deforestation, the relative attribution of the exposure to specific companies and international markets remains stable (Figure S13-16). Given the consistent patterns across datasets, uncertainty in remote sensing products should not be a barrier to corporate and government action on cocoa-driven deforestation. Yet, current approaches to attributing deforestation to cocoa and other crops, which intersect cocoa and forest maps developed in isolation, could be improved by developing harmonised land use products which simultaneously map multiple land covers and uses, and transitions over time, as Mapbiomas does in South America and Indonesia (Alencar *et al* 2020). Further, when attributing deforestation exposure to international markets, it is noteworthy that our results cover the country of import, rather than consumption. Consumption-based attribution studies, on the other hand, also introduce further methodological assumptions and modelling uncertainties and are agnostic as to the identity of which companies handle products along the supply chain (West *et al* 2022). Finally, though deforestation happens on farms, we map sourcing and deforestation

²⁰ Referring to ETH Zürich University and the Kalischek et al (2022) cocoa map.

exposure at the department-level, which is the finest resolution possible in the absence of publicly-available data about the location of farms supplying each company.

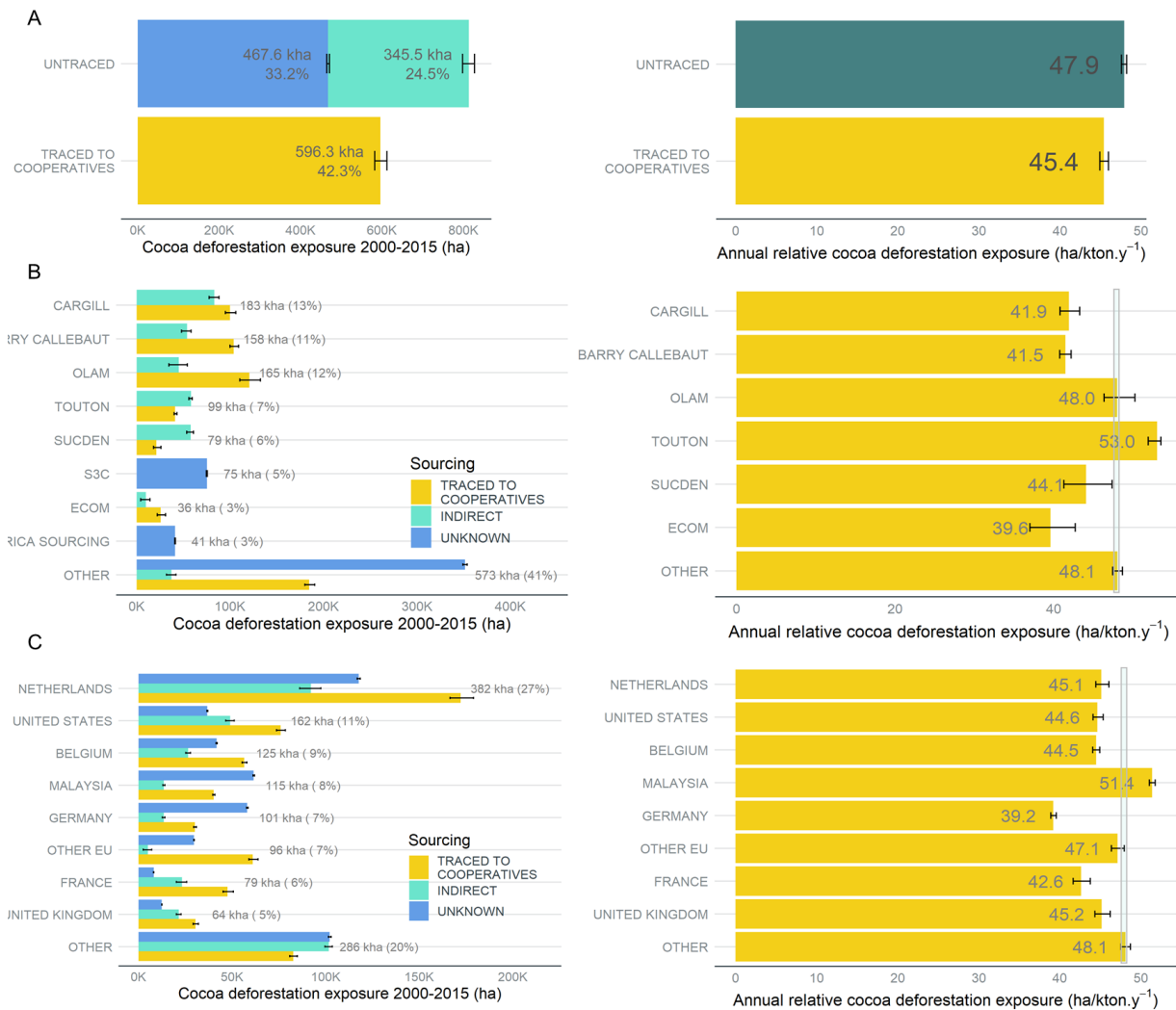


Figure 2.5. Cocoa deforestation exposure (CDE) in the cocoa supply chain. Comparison of sourcing traced to cooperatives, indirect and unknown sourcing in absolute (left) and relative (right) exposure, for the national exports (A), per trader (B) and per importing country (C). The CDE is the share of cocoa deforestation (2000 to 2015) attributed to the importing country or to the trader based on the volumes and the origin of the cocoa they purchased in 2019. The annual relative cocoa deforestation exposure in yellow is the CDE weighted by the sourced volume traced to cooperatives and annualised (divided by 15). The vertical line represents the annual relative CDE for untraced sourcing. Error bars represent the 95% confidence interval of the estimates.

2.3.4. Prospects for curbing cocoa-driven deforestation

Companies, as well as the forthcoming EU due-diligence legislation, count on farm-level traceability to reach zero-deforestation supply chains. If traceability offers a unique opportunity to improve the living conditions of farmers, through better traced payments and a clearer cooperative system (INADES *et al* 2022), several loopholes may undermine its effectiveness in curbing deforestation.

Company-led traceability is challenging and could be improved, but is not sufficient to address deforestation. Large parts of the supply chain are not mapped; only 44% of the Ivorian cocoa is traced to cooperatives, and 22% is traced to farm under the CFI. Companies in the CFI reported lower traceability in 2021 (72%) vs 2020 (74%) in their ‘direct’ supply chain from cooperatives. This decline highlights the challenge of traceability, not least the year-on-year changes in farmer membership of cooperatives and company programs requiring frequent updates of the supply base data (Barry Callebaut 2020). As company-led traceability only covers a partial share of the whole supply chain, it leaves gaps in which deforestation is left unmonitored. Buyers less exposed to reputational risks and less committed to sustainability can continue sourcing from these regions – and their non-compliant cocoa can end up in the indirect sourcing of committed actors. Similarly, as the EU due-diligence only applies to volumes sold in the EU, there is a risk of market bifurcation, with non-compliant cocoa sold to non-EU markets (40% of exports). Considering the market dominance of the EU and the similar due-diligence legislations under development in other importing markets, this risk of leakage is limited compared to other contexts, such as the Brazilian cattle sector (le Polain de Waroux *et al* 2019, zu Ermgassen *et al* 2020), but still remains. If traceability does not cover the whole supply chain, deforestation leakages risk undermining its conservation additionality (Gardner *et al* 2019, Garrett *et al* 2019, Gollnow *et al* 2022, Lambin *et al* 2018).

High hopes are placed in national farm-level traceability systems, as currently under development by the Coffee and Cocoa Council (CCC) in Côte d’Ivoire (Aboa 2022), **but they remain challenging and have to be transparent to be effective.** First, cooperatives play a crucial role in traceability but they are poorly structured (el Makhoulfi *et al* 2018). Many are the result of traders or local leaders’ business interest, with a membership ‘on paper’, rather than farmers deciding to work together (Ruf *et al* 2019). As a result, links between farmers and buyers are fluid with farmers selling to multiple buyers to cope with delays in payments (BASIC and FAO 2020, el Makhoulfi *et al* 2018, Nitidae and EFI 2021) and cooperatives buying from non-member farmers to meet the needs of their clients

(IDEF 2021, Stoop *et al* 2021). As much of Côte d'Ivoire's forest were cleared before the cut-off date proposed by the Commission (December 31, 2020), with less than 10% of the territory covered by tropical moist forest in 2019, the due-diligence's traceability requirements will be the major barrier to access the EU's market for Ivorian farmers and cocoa traders. Support from government and trading companies is thus required to improve the governance of cooperatives at the benefit of farmers and farm workers (Meemken *et al* 2019). Second, as non-compliant cocoa is widespread, it is crucial that the national traceability system is externally verifiable, and thus transparent. At least one-fourth of cocoa is produced within protected areas, and without transparency there is a risk that this cocoa continues to be hidden, jeopardising sustainability initiatives' impact (Gardner *et al* 2019). Making this data public would also alleviate companies' concerns about disclosing their competition-sensitive data, allowing for greater collaboration. Further, equitable frameworks are required to address non-compliant flows, otherwise leading to the exclusion of farmers who will have no other option than selling at low price to intermediaries (Bakhtary *et al* 2020).

Pre-competitive collaboration among traders and chocolate companies, at the landscape level, is one of the most promising ways to increase the effectiveness of sustainable sourcing initiatives, by delivering consistent impact at scale (Bakhtary *et al* 2020, IDEF 2021, Sayer *et al* 2013, Wolosin 2016, Carodenuto 2019). Companies buy from the same landscapes (Figure 2.6C), but rarely collaborate and implement their separate sustainability programmes with cooperatives that are part of their 'direct' sourcing – ignoring regions where untraced sourcing dominates (SI Figure S7). This lack of collaboration translates into scattered projects sprinkled over the territory, and into activities sometimes overlapping and redundant. For example, certification – which has been a major sustainability strategy in the sector for the past decade (Thorlakson 2018), led to considerable duplication of effort, with many double or triple certification of cooperatives (SI Figure S17). Furthermore, a lot happens outside companies' supply chains, with about half of deforestation in Côte d'Ivoire not directly linked to cocoa. Yet, as cocoa occupies large parts of agricultural land (Kalischek *et al* 2022), it drives deforestation indirectly by pushing other crops inside protected areas (Ajagun *et al* 2022, Kumeh *et al* 2022). To address deforestation, cocoa companies need to work beyond their individual supply chains. Landscape or jurisdictional approaches, defined as governance initiatives promoting sustainable resource use in a specific landscape (e.g. jurisdiction) through a formalised collaboration between all stakeholders (von Essen and Lambin 2021, Reed *et al* 2016), have the potential to internalise indirect sourcing and deforestation drivers beyond the focal commodity

(zu Ermgassen *et al* 2022). The CFI identified priority regions to establish multi-stakeholder landscape governance (CFI 2018), but progress is elusive with only a few pilot projects initiated (CFI 2022). The current CFI regions only capture 23% of the 2000-2019 cocoa deforestation and degradation and 32% of the remaining undisturbed TMF – these numbers could increase by 10% if other key departments were included (Figure 2.6A-B). Greater effort is needed, requiring government institutions to build the necessary enabling environment and companies to collaborate, support broader regulatory efforts, and provide funding beyond their own projects, focusing on landscapes most at risk of deforestation.

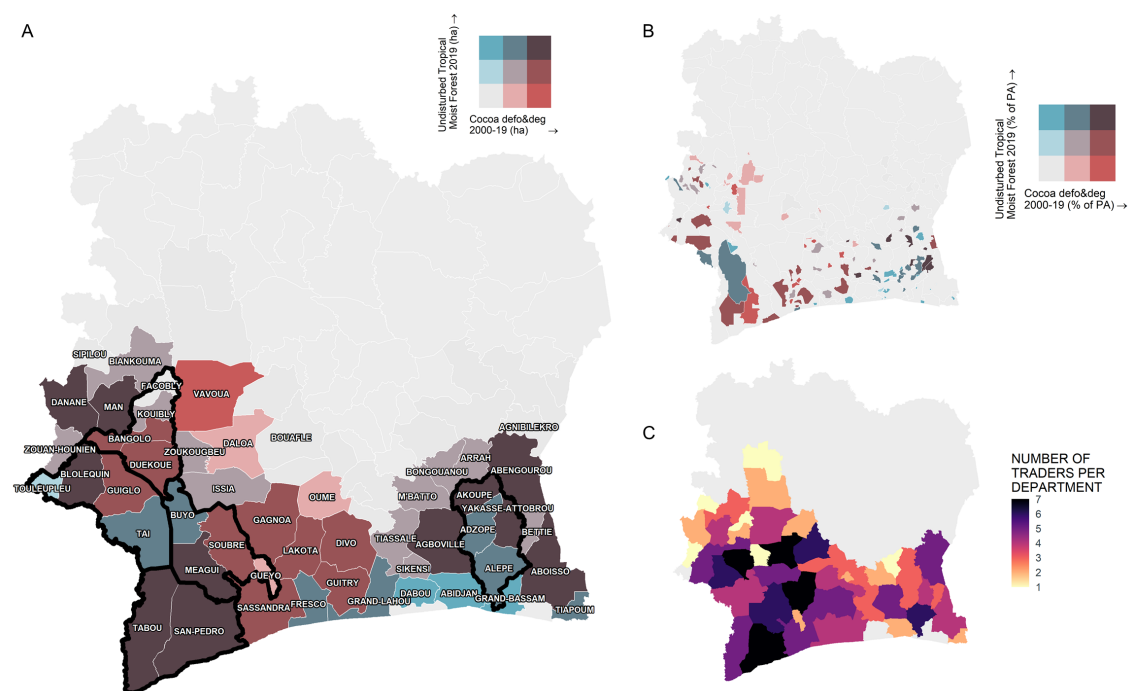


Figure 2.6. Critical departments and protected areas. (A) Bivariate map per department (with > 200 ha cocoa deforestation & degradation), with the remaining undisturbed Tropical Moist Forest (TMF) in 2019 (blue vertical scale) and the 2000 - 2019 cocoa deforestation & degradation (defo & deg) (red horizontal scale) in hectares, highlighting key departments which could be considered as priority landscapes (i.e. classified as 3-3 (dark brown) and 2-3 (dark blue)). The breaks of the color scales correspond to quantiles and are as follows for cocoa defo & deg: [202, 12 000] (12 000, 40 400] (40 400, 193 000] and for undisturbed TMF: [99, 2 100] (2 100, 15 500] (15 500, 292 000]. CFI priority regions are outlined in black. (B) Bivariate map per protected area (PA) (including classified forests, with > 0.005% undisturbed TMF in 2019 and cocoa defo & deg), with the remaining undisturbed TMF in 2019 (blue vertical scale) and the 2000 - 2019 cocoa defo & deg (red horizontal scale) in percentage of the PA area, highlighting key PA (i.e. classified as 3-3 (dark brown) and 2-3 (dark blue)). The breaks of the color scales correspond to quantiles and are as follows for cocoa defo & deg: [0, 26%] (26%, 58%] (58%, 92%], and for undisturbed TMF: [0, 2%] (2%, 13%] (13%, 98%]. (C) Number of traders with sourcing traced to cooperatives (in 2019) per department.

Beyond Côte d'Ivoire's borders, the above recommendations – i.e. implementing landscape approaches, regulatory policies, and a transparent national traceability

system – also apply to highly forested countries, where cocoa is expanding (Sassen et al 2022).

2.4. Conclusion

Our findings show that the majority of cocoa exports from Côte d’Ivoire are either untransparent (sourced via traders that don’t disclose any information) or indirectly sourced, with the origin unknown. Companies do not know where this indirectly sourced cocoa originates from and are therefore incapable of evaluating if it is tied to sustainability issues, including deforestation or child labor, and can thus hardly act directly to improve them.

Our results also demonstrate the role of cocoa as a major driver of deforestation in Côte d’Ivoire, with almost half of the undisturbed tropical moist forest lost between 2000 and 2019 converted into cocoa. Cocoa deforestation exposure varies among traders and destinations, but is fundamentally embedded in all the supply chain. Almost 60% of this deforestation exposure is linked to untraced sourcing, highlighting the crucial importance of incorporating indirect sourcing in zero deforestation commitments.

In the absence of strongly enforced land use policies, full farm-level traceability, as required by the forthcoming EU due-diligence regulation, is a prerequisite for zero-deforestation commodities. Yet, companies’ traceability cannot be considered as a stand-alone solution to deforestation; if not implemented by the whole industry, it leaves gaps that are likely to cancel out its expected impact on deforestation. Furthermore, without transparency, committed actors cannot be held accountable. Transparent national traceability systems, coupled with a robust deforestation monitoring system, can thus be an essential stepping-stone to curb deforestation, but must be combined with land use policies, landscape initiatives, and increased means targeting remaining forests to ensure effective forest conservation.

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Chapter 3

Direct and indirect deforestation for cocoa in the tropical moist forests of Ghana

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Abstract

Across the tropics, cocoa is one of the main drivers of deforestation. In Ghana, the world's second largest cocoa producer, the role of each of the main economic sectors in driving deforestation remains, however, contested – with cocoa, mining, logging, and plantations each blaming the others. Previous work has also suggested that food crops are displaced into forests by cocoa expansion, raising concerns about indirect land use change and impacts on food availability. Here, using satellite-based maps and secondary data, we quantify the *direct* deforestation and forest degradation between 2000 and 2019 in the entire cocoa-growing region of Ghana which is attributed to the land uses detected in 2019. Then, we use a land-balance approach to assess the *indirect* role of the expansion of land uses in deforestation and degradation for food crops. We find that cocoa was the major direct driver over that period (~57%), followed by food crops and logging (~39% in total), then by mining (3%), tree plantations (2%) and settlements (<1%). Roughly 30% of the deforestation and degradation of land used for food crops in 2019 was attributed to their displacement from elsewhere by other land uses – with 15% (9%-17%) due to cocoa expansion into food crops. In cocoa-saturated regions, indirect deforestation is likely to increase as forests only remain in gazetted areas, where growing food crops is either legal or more tolerated by officials than cocoa. Accounting for its direct and indirect role, cocoa was associated with ~20% of the deforestation in gazetted areas. This research highlights the need to move away from sustainability efforts targeting one single commodity at a time to more transformative approaches that develop a coordinated vision across the food and land system. Such a vision would regulate multiple commodities holistically, address lock-ins, and reconsider the overall use of land systems in the region (how much land should be dedicated to cocoa, how and for whose benefits).

3.1. Introduction

Cocoa production is one of the most important economic sectors in Western Africa, employing millions of smallholder farmers and workers. With over 800,000 smallholder farmers producing ~800,000 tonnes of cocoa beans yearly, Ghana is the second-largest cocoa producer in the world, accounting for about 20% of global cocoa production (FAO, 2022). After gold and crude petroleum, cocoa is the third most important export of Ghana, worth \$2.4 billion annually, or more than 17% of Ghana's total export earnings (OEC, 2021).

Yet, the crucial importance of cocoa in the Ghanaian economy comes at a cost: Ghana lost over 60% of its forest cover from 1950 to the turn of the last century (Forestry Commission, 2016). The REDD+ Strategy of Ghana (2016) states that the country has one of the highest deforestation rates in the world with an annual rate of 2% (i.e. 135,000 ha/year). Although cocoa is widely discussed as a major driver of deforestation (Goldman et al., 2020; Pendrill et al., 2019), its precise contribution to deforestation in Ghana remains contested. Pan-tropical or pan-African studies provide varying estimates of cocoa-driven deforestation, attributing tree cover loss to cocoa using national-level agricultural statistics (Pendrill et al., 2019), coarse land use data (Goldman et al., 2020), high-resolution remote sensing (Masolele et al., 2024), or a mix of remote sensing data and agricultural statistics (Singh & Persson, 2024) (SI table S1).

Aside from cocoa, other drivers are put forward as having a major role in the deforestation of the Ghanaian tropical moist forest, such as illegal artisanal mining (*galamsey*), legal and illegal timber exploitation, tree plantations (oil palm, rubber, coconut), urban sprawl, or food crops, each of the sectors involved laying blame on the others (Yoda, 2019). Food crops were for instance identified as the main driver of deforestation between 2019 and 2022 across the six priority cocoa landscapes in Ghana (CFI, 2023). Mining has also been recently flagged as expanding not only into forests but also into cocoa plantations, threatening cocoa production (Onah, 2023).

Beyond driving deforestation directly by expanding into forests, these various land uses can cause deforestation indirectly by competing and expanding into other land uses that are themselves displaced into forests (Kuschnig et al., 2021; Meyfroidt et al., 2013, 2018). Indirect land use change (iLUC) is a form of spillover where land use change in one place is caused by land use change in another place (Meyfroidt et al., 2020). These complex land use dynamics depend on the competitiveness of different crops and land uses, land suitability, opportunity costs, and policies (Meyfroidt et al., 2018; Turner et al., 2020). Studies in some regions of Ghana

show that cocoa expansion is taking up a considerable share of available farmland and is linked to food crops being displaced in neighbouring forests, raising concerns about food availability for households converting their food crops into cocoa plantations (Ajagun et al., 2022; Kumeh et al., 2022). Food crops (mainly plantain, cocoyam and yam) are indeed often intercropped with tree crop seedlings, and are then displaced once the canopy closes (Acheampong et al., 2019; Asubonteng et al., 2018).

In response to civil society pressure, chocolate companies have made a series of zero deforestation commitments and investments in traceability, seeking to identify the origin of the cocoa they buy. The urgency behind these efforts has been further increased by the European Union's deforestation-free regulation (EUDR), requiring companies to demonstrate that imports of deforestation-risk products (such as cocoa or chocolate) do not originate from recently deforested nor degraded land (EC, 2021). However, the effectiveness of "narrow" zero-deforestation commitments or traceability efforts focussed on specific commodities as 'solutions' to deforestation is increasingly questioned (Gardner et al., 2019; Garrett et al., 2019; Gollnow et al., 2022; Lambin et al., 2018; Panwar et al., 2023; Renier et al., 2023), as they ignore the complex land use dynamics outlined above. Hence, even if supply chains are EUDR-compliant and companies claim deforestation-free sourcing, whether the cocoa sector can become free of all deforestation – direct and indirect – remains an open question.

A comprehensive picture of the land use dynamics and their impacts on forests across the Ghanaian cocoa-growing area remains lacking, as previous studies focus on specific regions (Acheampong et al., 2019; Ashiagbor et al., 2022; Asubonteng et al., 2018; Benefoh et al., 2018; Kyere-Boateng & Marek, 2021), do not disentangle the contribution of each crop in national agricultural deforestation (Acheampong et al., 2019; Forestry Commission, 2016), or only evaluate the land use following deforestation on a plot, without accounting for indirect land use dynamics (Masolele et al., 2024). Here, our objective is to better understand the role of cocoa and other land-intensive activities in driving pressure on land resources, and ultimately on forests, in the whole cocoa-growing region of Ghana (~8 Mha), by assessing their direct (i.e. deforestation attributed to the detected land use in 2019) and indirect deforestation footprints (i.e. deforestation of land used for food crops in 2019 and attributed to the expansion of another land use) over the period 2000-2019. We use existing Earth Observation and statistical datasets to assess direct deforestation, and a land balance approach to quantify indirect deforestation.

3.2. Methods

To calculate the direct and indirect forest footprint of land uses, we followed five main steps summarized here and developed further below (see Figure 3.1 for the summary of datasets and workflow, and Figure 3.2 for definitions). First, we quantified the direct deforestation and degradation footprint of cocoa, oil palm, rubber, coconut, mining, and urban sprawl using the most accurate and up-to-date satellite-based datasets available, thus covering the period 2000-2019 (section 3.2.2). Second, with the same data we calculated their expansion outside what was forest in 2000 (section 3.2.3.1). Third, absent a comprehensive food crop satellite-based dataset, we used statistical records and a land balance approach to estimate how much of that expansion occurred on food crops, thus displacing food crops elsewhere, based on the relative importance of the food crop area in the region compared to other pools of land ‘available’ for expansion (section 3.2.3.2.a). Fourth, following the same approach, we estimated the area of food crops which expanded or were displaced onto forests (section 3.2.3.2.b-c). Fifth, we attributed the total deforestation and degradation of land used for food crops in 2019 to the other land uses based on their relative expansion, to calculate their indirect footprint on forests through food crop displacement (section 3.2.3.3) (see SI Figure S1 for the detailed steps of the land balance). We performed the calculations at the region level, to capture the specific dynamics in each region, and then repeated the land balance and indirect deforestation attribution with data aggregated directly at the level of the whole study area.

3.2.1. Study area

The study area corresponds to the tropical moist forest zone of Ghana, called the High Forest Zone (HFZ), where cocoa is grown (Figure 3.1). This is a ~8 Mha area, located mainly in the Southwestern part of the country (W. Asante, 2014). We excluded the transition zone and the savannah region of the country as they have different land use dynamics, with different crops and deforestation drivers. To allow for a better understanding of local dynamics, the analysis was carried out at the level of six administrative regions rather than only at the HFZ level. To facilitate interpretation, in this work the ‘Western’ region refers to the previous region’s delineation (pre-2020) and thus includes the current (2024) Western-North region, while the ‘Brong Ahafo’ region refers to the current Ahafo region plus a part of the new Bono region, and the ‘Volta’ region includes districts of the current Oti region. We restricted Brong Ahafo, Ashanti and Volta regions to the HFZ by removing 43 districts located beyond the delineation of the HFZ

(hereafter, ‘regions’ thus refer to these trimmed regions, not including any district beyond HFZ) (SI Figure S2).

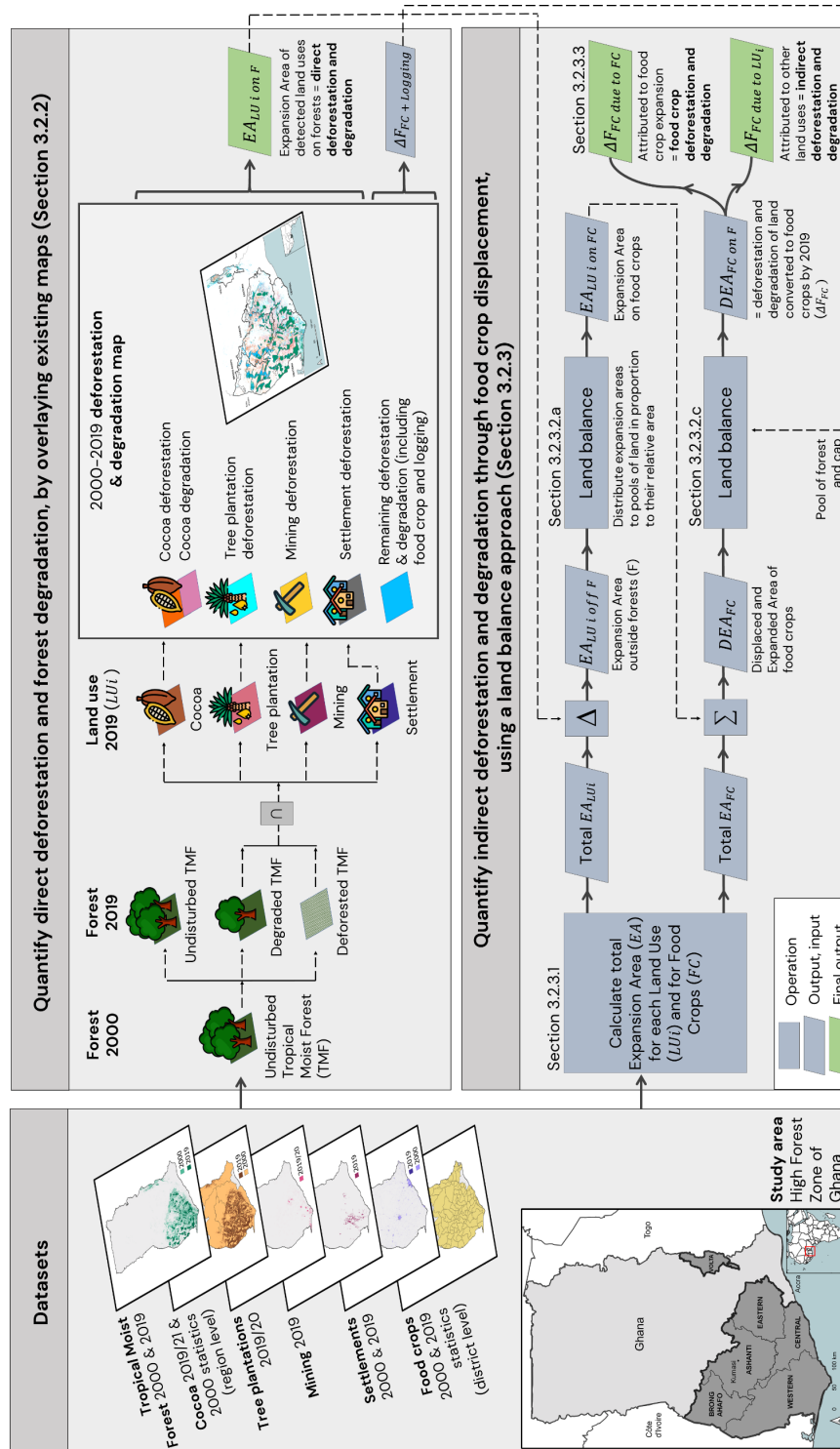


Figure 3.1. Study area, datasets and workflow. The top panel presents the steps of the production of the *direct* deforestation and degradation figures and map over 2000-2019 (cf. Figure 3.4A), and the bottom panel describes the steps to quantify the amount of deforestation and degradation of land used for food crops in 2019 for which the establishment of food crops is attributed to their displacement from elsewhere by other land uses (i.e. *indirect* deforestation and degradation).

3.2.2. Quantifying direct deforestation and forest degradation

To quantify the *direct* deforestation and degradation of the main land uses, in the absence of a publicly accessible national land cover map, we overlaid several remote-sensing maps with a map of Tropical Moist Forest (TMF) produced by European Commission’s Joint Research Centre (JRC), depicting annually from 1990 to 2020 the extent of undisturbed tropical moist forest and its changes at 30 m resolution (Vancutsem et al., 2021).

Due to the limitation in the data (land use maps being available for the year ~2019 only, not as annual time series), we measured deforestation and degradation that occurred between 2000 and 2019, and we first attributed the change to the observable land use at the end of the period. Thus, in the context of this paper, “direct” deforestation and degradation refers to land detected as undisturbed forest in 2000 converted to one of the considered land uses detected by satellite imagery in 2019. Therefore, our approach does not capture the possible intermediate land use transitions – yet, this period covers ~20 years, which is the IPCC default inventory period for land use change attribution (Maciel et al., 2022) and which we consider to be a reasonable timeframe to capture the major forces driving land use dynamics over that period. In the HFZ, food crops are often intercropped with cocoa seedlings in the first years of cocoa plantation establishment after deforestation (Acheampong et al., 2019; Asubonteng et al., 2018), but in the context of the 2000s’ cocoa boom and the comparative perceived advantages of cocoa, there is convincing evidence to consider that land where cocoa is detected in 2019 was initially deforested with the intention of growing cocoa – even if food crops were present at the outset (Clough et al., 2009; Ruf & Siswoputranto, 1995) (text S1.1).

The maps overlaid with the TMF map include i) a **cocoa land use map** for the years 2019–2021 at 10 m resolution produced for Ghana and Côte d’Ivoire (Kalischek et al., 2023) (hereafter, the ‘2019 cocoa map’); ii) a **global tree plantations map** for the year 2019–2020 (Vancutsem et al., 2021) which includes rubber and oil palm for 2019 at 10 m resolution (Descals et al., 2021) and a global closed-canopy coconut map for 2020 at 20m resolution (Descals et al., 2023) (both cocoa and coconut being perennial crops, limited year-on-year variation is expected); iii) a **global mining land use map** for the year 2019 produced at 10 m resolution, which includes large-scale, artisanal and small-scale mining, formal and informal, and covering all ground features related to mining, e.g. open cuts, tailing dams, waste rock dumps, water ponds, processing infrastructure (hereafter, the ‘2019 mining map’) (Maus et al., 2020); (iv) a **global settlement footprint map** for the years 2000 (at 30 m resolution) and 2019 (at 10 m resolution) (Marconcini et al.,

2021). These spatial datasets inherently contain detection errors, but their accuracy spans from 84% to 99%, making them suitable for such an analysis (see SI table S2 for table of datasets and their accuracy, and table S3 and text S1.2 for overlaps).

We created a map of the land-use change between 2000 and 2019 including deforestation and degradation, their association with land uses, and the remaining undisturbed and degraded TMF cover. A degraded forest is defined by the JRC TMF dataset as a forest that has been temporarily disturbed (<2.5 years), excluding tree plantations (text S1.3). Here, ‘deforestation’ corresponds to pixels that switched from undisturbed TMF to another land cover between 2000 and 2019, with at least 2.5 years of disturbance, and ‘degradation’ to pixels that switched from undisturbed to degraded TMF. We kept this distinction for pixels detected as cocoa in 2019 – which were classified as ‘cocoa deforestation’ and ‘cocoa degradation’. Cocoa degradation could be forest converted to cocoa but with a relatively dense remaining shade canopy cover, or cocoa plantations that were too recently established to be classified as deforestation (having less than 2.5 years of detected disturbance). Young cocoa farms are, to some extent, not detected by the cocoa map – thus very recent deforestation for cocoa is likely classified as non-cocoa deforestation or degradation. For the other land uses, i.e. mining, tree plantations and settlements, we merged degradation with deforestation (thus called ‘mining deforestation’, etc) because the areas of degraded TMF detected as one of these land uses were very limited compared to cocoa, and these activities are expected to lead to a full removal of the natural forest.

Due to the low quality of clouded satellite images of the HFZ of Ghana in the 1990’s, highly shaded cocoa farms in 1990 (the starting point of the JRC TMF time series) may have been classified as undisturbed TMF – meaning that the removal of shade trees or the rehabilitation of old abandoned cocoa farms may be categorised as cocoa deforestation or degradation. To mitigate this and limit the misclassification of cocoa as forest, we filtered the pixels classified as undisturbed TMF, keeping those passing two thresholds: i) vegetation must be > 8 m in height (based on two global forest height datasets for 2000 (at 30 m resolution) and 2020 (at 10 m resolution) (Lang et al., 2023; Potapov et al., 2022)), and ii) pixels must form a minimum of 1 ha patch to be considered as forest. These restrictions decreased the detected undisturbed forest in 2000 from 4.7 Mha to 2.6 Mha, which is similar to the 2000 HFZ forest area estimate of 2.7 Mha by the Forestry Commission (2021, p124-127).

For each remotely-sensed land use (LU), we calculated the forest area that they replaced, thereby quantifying their direct role in deforestation and forest degradation over the 2000-2019 period (Eq. 1) (SI Figure S1B).

$$EA_{LU\ i\ on\ F} = F_{y1} \cap LU_{iy2}, \quad (\text{Eq. 1})$$

With $EA_{LU\ i\ on\ F}$ the expansion area of LU i on forest (F_{y1}).

Deforestation or forest degradation detected by the TMF dataset but without detected post-clearing LU can correspond to logging, fuelwood collection, or clearing for food crops – for which no robust satellite-based dataset was available in the HFZ of Ghana (SI Figure S3). Hence, once all other quantifiable drivers are subtracted from the total area of deforestation and degradation, we classify the remainder as the deforestation and degradation linked to food crops or logging / fuelwood. We analysed the dynamics of direct deforestation drivers across distinct types of gazetted and non-gazetted land using data from Resource Management Support Centre of Ghana.

3.2.3. Quantifying indirect deforestation and degradation through food crop displacement

3.2.3.1. Estimating the total expansion of land uses and food crops

To estimate the deforestation and degradation of land used for food crops in 2019 which was due to (i) the net expansion of food crops itself (linked to underlying drivers such as demographic growth) versus (ii) the expansion of other land uses that displace food crops into forest, we first need to calculate the total food crop net expansion (i.e. the difference between 2000 and 2019), as well as the area of food crops displaced by the expansion of other land uses. For the latter, we need to estimate the total expansion of these other land uses outside forests.

For estimating the net food crop expansion, in the absence of a remote-sensing dataset, we used statistical data of the land area farmed for a list of crops identified as food crops per district in 2000 and 2019, obtained from the Ghana Statistical Service.

The expansion area of each land use *outside forests* was obtained by subtracting the area of expansion into forest (i.e. direct deforestation) from the total expansion of each land use. To obtain the latter, we used different methods according to the availability of data for each land use (see table S4). For settlements, we subtracted the extent of the land use in 2000 from its extent in 2019, both observed in satellite data. For mining and tree plantations, we lacked data on the extent in 2000. Hence,

we used the total 2019 area as the upper-bar expansion area (i.e. considering that all mining sites and tree plantations detected in 2019 were established after 2000). Conversely, we used the 2019 mining and tree plantation area detected as forest in 2000 as a lower-bar estimate (i.e. considering only the area directly replacing forest over 2000-2019 as new, and thus that all the area outside forest was established before 2000). Finally, for cocoa, we estimated the area in 2000 by dividing the cocoa production per region (COCOBOD, 2023) averaged over 1999-2001 by the yields collected in 2001/2002 in five of the six cocoa regions (Ruf, 2007b) (SI text S1.4). To account for uncertainty in yields, we computed 10% upper- and lower-bar estimates (SI table S5). For the sixth region, Volta, we imputed missing yield information from the other regions' average. The yield estimates were collected for productive cocoa plantations, yet, the total area of cocoa plantations also includes old plantations not harvested anymore as well as young, not yet productive areas. To account for these unproductive cocoa areas, the area was increased by 24%, as assessed for 2001/02 in Ashanti, Brong Ahafo and Western regions by Teal et al. (2006). The expansion of cocoa was thus calculated as the difference between the satellite-based area in 2019 and the estimated cocoa area for 2000. The upper- and lower-bar estimates of the total expansion of mining, tree plantations and cocoa reflect the various data uncertainties and serve as robustness checks (referred to as 'max' and 'min' land use expansion scenarios).

3.2.3.2. Estimating food crop displacement

a. Expansion of land uses on food crops

To estimate the area of food crops displaced by other land uses, we used a land balance approach similar to Pendrill et al. (2019) and the World Business Council for Sustainable Development official guidance for land use change accounting (WRI & WBCSD, 2022). Land balance is used as a way of accounting for overall land use expansion including indirect land use change and deforestation. The following analyses were conducted at two scales: the region level, with results of each region then summed to get the total at HFZ level, and directly with data aggregated at the entire HFZ level, in order to identify dynamics at different scales (zu Ermgassen et al., 2024).

First, we estimated how much cocoa, tree plantations, mining, and settlement land uses expanded on food crops over the 2000-2019 period (SI Figure S1C). The total expansion calculated above may have occurred on various 'pools' of land: i)

forests (F); ii) ‘other’ land (OL) not classified in one of our LULC categories²¹; iii) food crops (FC); and, for land uses other than cocoa, iv) abandoned cocoa farms ($CC_{converted}$)²².

Satellite data provides the direct expansion of land uses on forests (Eq. 1) but no spatially-explicit data exist for the expansion which occurred outside forests, neither a robust quantification of the proportion of each land use expansion on specific land pools. We thus reconstructed the amount of expansion of LULC on those outside-forests landpools ($Pool\ j$) by assuming that these LULC would expand without preferential bias towards any specific pool. We thus distributed the expansion area of LU i outside forests ($EA_{LU\ i\ off\ F}$) to each pool in proportion to the relative area of these pools within the region (Eq. 2-3).

$$EA_{LU\ i\ on\ Pool\ j} = EA_{LU\ i\ off\ F} * \frac{Area\ pool\ j}{\sum_j Area\ pool\ j}, \text{ (Eq. 2)}$$

With

$$EA_{LU\ i\ off\ F} = EA_{LU\ i} - EA_{LU\ i\ on\ F}, \text{ (Eq. 3)}$$

Each result was capped so that the expansion cannot exceed the overall pool area – any expansion in excess was redistributed to the other pools using the same method.

b. Amount of food crop displacement and expansion

The total expansion of all other land uses on food crops ($\sum_i EA_{LU\ i\ on\ FC}$) was taken as an estimate of the area of food crops displaced, and which was relocated elsewhere²³. This area was combined to the net expansion of food crops (i.e. new demand for food crops) to obtain the total displaced and expanded area of food crops (DEA_{FC} , Eq. 4).

²¹ e.g. fallows, bush, woodland that are <1ha or <8m in height or, for regions fringing the transition zone, other land uses such as orchards or cashew plantations.

²² Indeed, two regions (Eastern and Ashanti) have seen their net cocoa area *outside* forests contracting over 2000-2019 - i.e. cocoa may have expanded on forests, but the area of cocoa already present outside forests in 2000 decreased, thus freeing space for other land uses. This contraction area is used as the potential pool of cocoa land that can be converted in iv). As we only look at net expansion of land use, we do not account for new cocoa expanding into old cocoa (because this does not create any net change).

²³ In theory, pressure on food crop area could induce intensification, reducing the amount of new cropland area needed to substitute for the area displaced, but we could not find any study or evidence suggesting that farmers are increasing their inputs (labour, capital inputs) on food crops for intensifying their cultivation as a result of land pressure. Rather, farmers tend to focus their limited amount of inputs on cash crops (KI6,7). Alternatively, though not accounted for either, converted food crops could also not be entirely displaced, either resulting in more food imports or worsened food security, or could lead to an increased food crop area if displaced on marginal land.

$$DEA_{FC} = EA_{FC} + \sum_i EA_{LU\ i\ on\ FC} , \text{ (Eq. 4)}$$

c. Distribution of food crop displacement and expansion

Similarly to the LU expansion shared among various pools of land, this area of food crop displacement and expansion was then distributed to i) forests; ii) ‘other’ land; and iii) converted cocoa, based on their relative area in the region (Eq. 5) (SI Figure S1D)²⁴. We assumed that food crops do not replace the commercial LU, which are more profitable (see section 3.4.2). The ‘pool’ of forest available for food crops was the total 2000 forest area, minus the forest replaced by the other land uses (i.e. these land uses are considered as having priority over food crops), and capped by the ‘remaining’ deforestation and degradation, i.e. deforestation and degradation which was not attributed to any other land use and that can thus be due to food crops. The pool of ‘other’ land on which food crops may have expanded or have been displaced is the total land area minus the land area occupied by all other LU (either already in 2000 or on which they expanded up to 2019), and minus the food crop area of 2000. Finally, the pool of converted cocoa is the same as above, minus the area onto which LU other than cocoa are considered as having expanded.

$$DEA_{FC\ on\ Pool\ j} = DEA_{FC} * \frac{Area\ pool\ j - EA_{LU\ i\ on\ Pool\ j}}{\sum_j Area\ pool\ j - EA_{LU\ i\ on\ Pool\ j}} , \text{ (Eq. 5)}$$

Again, each result was capped so that the expansion in each pool does not exceed this pool’s area. This cap was reached for food crop expansion into deforested areas (thus exceeding the area of "remaining deforestation and degradation") in Central Region (0-3 Kha excess depending on the expansion scenario) and Western Region (10 Kha excess in the upper-bar expansion scenario, redistributed to the other pools).

3.2.3.3. Attributing the deforestation and degradation of land used for food crops in 2019 to the expansion of other land uses

The total deforestation and degradation of land used for food crops in 2019 – which corresponds to the total displaced and expanded area of food crops into forests ($DEA_{FC\ on\ F}$) – was then attributed to food crop expansion itself (EA_{FC}) (linked to underlying drivers such as population growth) versus displacement due to other land uses. The share of the former corresponds to the net expansion of food crops (EA_{FC}) relative to the total DEA_{FC} (Eq. 6) (SI Figure S1E). The share of each other

²⁴ As an example, in the Western Region where there was 812 Kha of forest and 243 Kha of ‘other’ land not converted to other land uses, and no converted cocoa, 77% of the food crop displacement and expansion is considered having moved into forests (812/(812+243+0)).

land use in indirectly driving the remaining food crop deforestation ($\Delta F_{FC \text{ due to } LU_i}$) was based on their relative expansion on food crops as compared to the total displaced and expanded area of food crops (Eq. 7).

$$\Delta F_{FC \text{ due to } FC} = DEA_{FC \text{ on } F} * \frac{EA_{FC}}{DEA_{FC}}, \text{ (Eq. 6)}$$

$$\Delta F_{FC \text{ due to } LU_i} = DEA_{FC \text{ on } F} * \frac{EA_{LU_i \text{ on } FC}}{DEA_{FC}}, \text{ (Eq. 7)}$$

We then summed the direct and indirect deforestation and degradation areas attributed to each driver over 2000-2019.

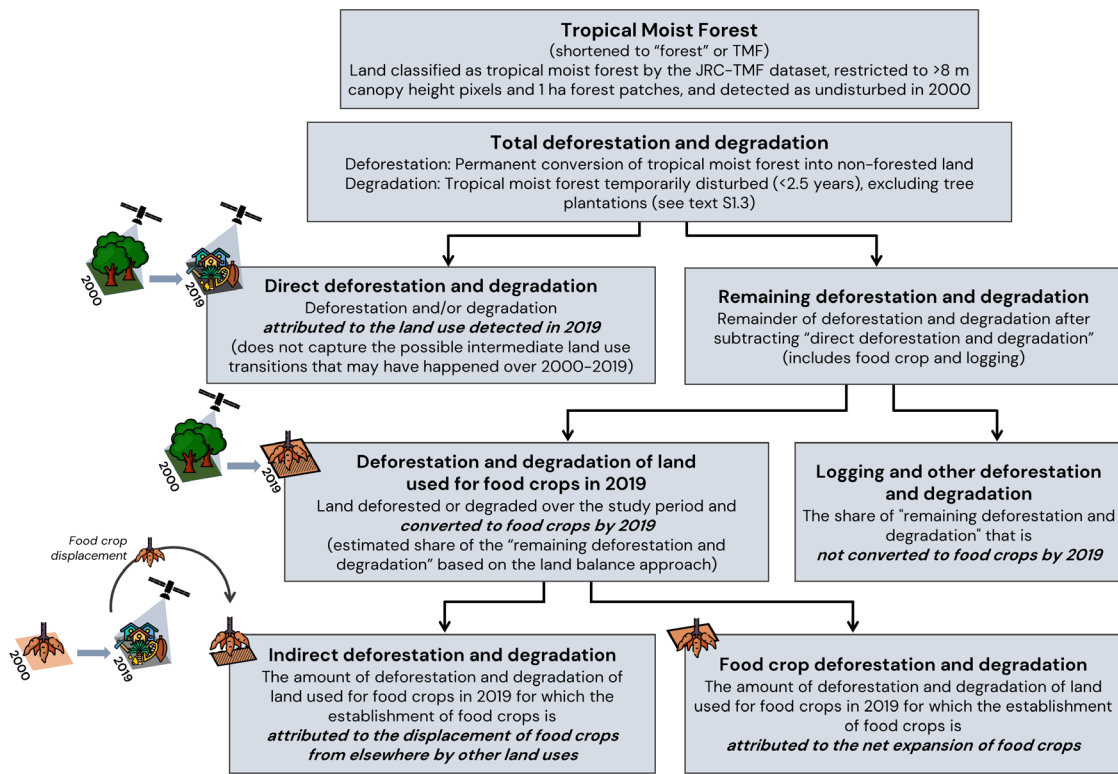


Figure 3.2. Land use change definitions. "Direct" in the context of this paper is used in contrast with "indirect": "direct" here means that there is a direct spatial association between deforestation and the detected land use, i.e., they both occur at the same location. "Indirect", in contrast, means that they are linked but do not co-occur spatially. The term "direct" is however not used for food crops to distinguish them from land uses observed by Earth Observation data. For deforestation where the land use inferred in 2019 is food crops, we use "Deforestation and degradation of land used for food crops in 2019", and for the share of that that was attributed to the net food crop expansion, we use "Food crop deforestation and degradation", as opposed to the share attributed to the expansion of other land uses that we call "Indirect deforestation and degradation" (i.e., food crops in 2000 converted to one of these other land uses by 2019 and displaced in forests) and which, in our research, only refers to the displacement of food crops (no displacement of other land uses are considered).

3.2.4. Validation of assumptions with key informants

To complement the literature review, we conducted semi-structured interviews with a set of 18 key informants (KI) (cocoa farmers, community leaders, NGOs, academics, government officials), to analyse the main dynamics in food crop displacement and cocoa deforestation and to cross-check the main assumptions of our methods (e.g. yields and uncertainty range of cocoa productivity in 2000, percentage of unproductive cocoa, prevalence of logging, typical land use trajectories and priorities of certain land uses over others (e.g., cash crop over food crops)) (SI text S1.5, table S6).

3.3. Results

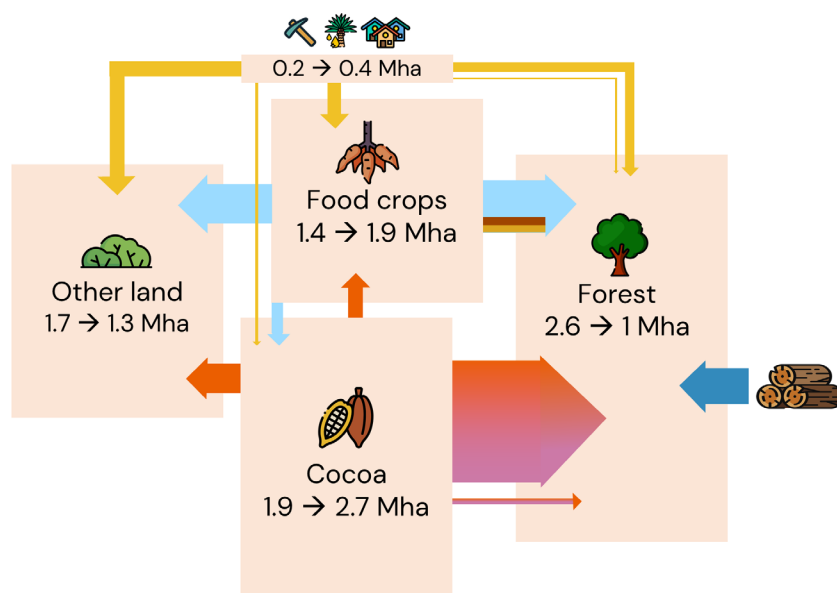
3.3.1. Direct and indirect drivers of deforestation and degradation

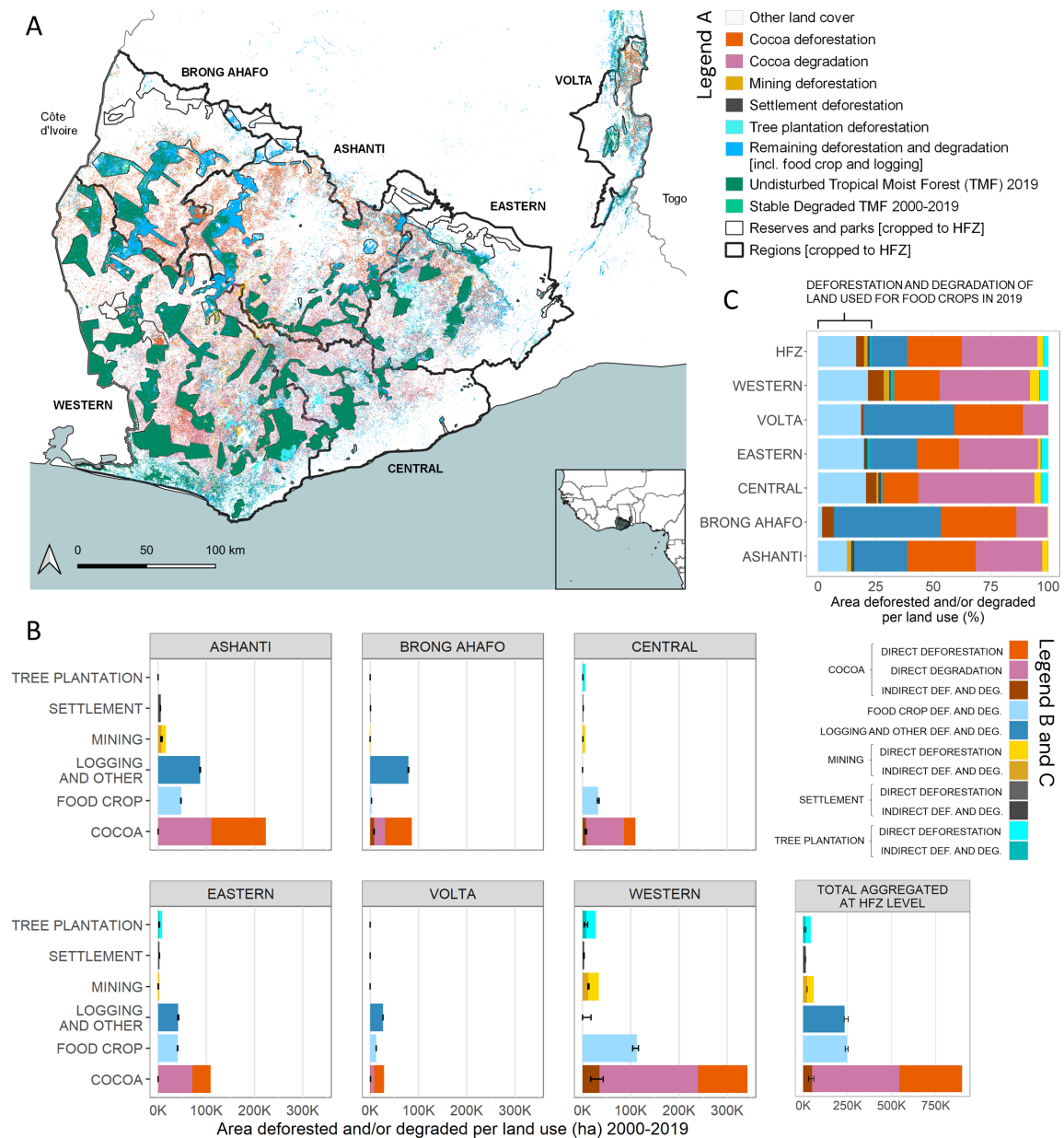
Over the 2000-2019 period, we found that, in the entire high forest zone (HFZ) of Ghana, 1.5 Mha (58%) of the initial 2.6 Mha of undisturbed tropical moist forest (TMF) was lost, either degraded or deforested, leaving the country with about 1 Mha of undisturbed TMF and 0.3 Mha of degraded TMF in 2019, representing respectively 14% and 4% of the HFZ (Figure 3.3, Figure 3.4A, SI table S7).

Cocoa replaced 848,000 ha of forest (354,000 ha deforested and 494,000 ha degraded), representing 57% of the total deforestation and degradation, making it the major direct driver of forest loss over this period. Mining (3% of the total deforestation and degradation), tree plantations (2%) and settlements (<1%) had, comparatively, a limited direct impact on forests over this period (Figure 3.4B-C, SI table S8) (“direct” drivers refer to land uses detected in 2019 on land deforested or degraded between 2000 and 2019).

We estimated that 23% (CI: 21.8%-23.3%) of the total deforestation and degradation was on land used for food crops in 2019 (348,000 ha, 327-350 Kha), separated into 17% (15.9%-17.0%) attributed to the net expansion of food crops itself, representing 249,000 ha (238-255 Kha), and 7% (4.8%-7.4%) (i.e. 99,000 ha, 72-112 Kha) coming from displacement (i.e., food crops driven out of a prior area by the expansion of other land uses). In total, 28% (22%-32%) of the forest converted to food crops was thus indirectly driven by other land uses – with 15% (9%-17%) due to cocoa, 6% (6.4%-6.6%) to mining, 4% (3.6%-4.4%) to settlements and 3% (1.9%-4.5%) to tree plantations (oil palm, rubber, coconut).

The remaining non-attributed deforestation and degradation (16% or 235,000 ha, 232-255 Kha) referred hereafter as ‘logging’, is considered to be mainly due to





3.3.2. Distinct on- and off-reserve deforestation and degradation patterns

In Ghana, forests are distinguished between those located in gazetted areas, comprising forest reserves (land often called “on-reserve”), national parks, and game reserves, and those ‘off-reserves’, i.e. non-gazetted areas. Unlike off-reserve land, gazetted areas are protected by legal provisions restricting all forms of agricultural and forestry activities, at the exception of forest reserves where some activities are allowed in specific portions with the necessary permits, such as timber exploitation (Forestry Commission, 1998, p22-29) and admitted farms, as well as food crops under the ‘Modified Taungya System’ (Acheampong et al., 2016).

Over 2000-2019, 79% of the total deforestation and degradation in the high forest zone happened off-reserve, where 75% of the undisturbed forest of 2000 was deforested and 11% degraded (SI table S7-8). Gazetted forests in the HFZ lost 26% of their 2000 undisturbed forest cover (314,000 ha of the initial 1.2 Mha, with 14% deforested and 12% degraded). 99% of this loss happened in forest reserves. Only 2074 ha were lost in the 5 national parks and game reserves of the HFZ (i.e. 3% of their 2000 undisturbed forest). In 2019, 83% of Ghana’s remaining 1 Mha of undisturbed tropical moist forest were located in protected forests (93% in forest reserves).

In gazetted areas (mostly forest reserves), 85% of the total deforestation and degradation is not directly attributed to any of the satellite-based land uses (264,000 ha – 41% deforestation, 44% degradation), against only 27% off-reserve. Only 12% of the deforestation and degradation is directly attributed to cocoa (38,000 ha), 2% to tree plantations (6,500 ha) and <1% to mining (2,800 ha). The main direct drivers of deforestation and degradation within forest reserves between 2000 and 2019 were thus food crops and logging (Figure 3.5), while cocoa was by far dominating off-reserve. Assuming a similar proportion of food crop (60%) vs. logging (40%) in the ‘remaining’ deforestation and degradation in forest reserves as in the entire HFZ, we estimated that ~51% of the total deforestation and degradation in forest reserves was on land used for food crops in 2019 (60% of 85%), with ~8% indirectly driven by cocoa (15% of 51%). Accounting for both its direct impact and indirect impact through food crops displacement, cocoa can be associated with at least 20% of the deforestation and degradation in the gazetted areas (12% directly, ~8% indirectly).

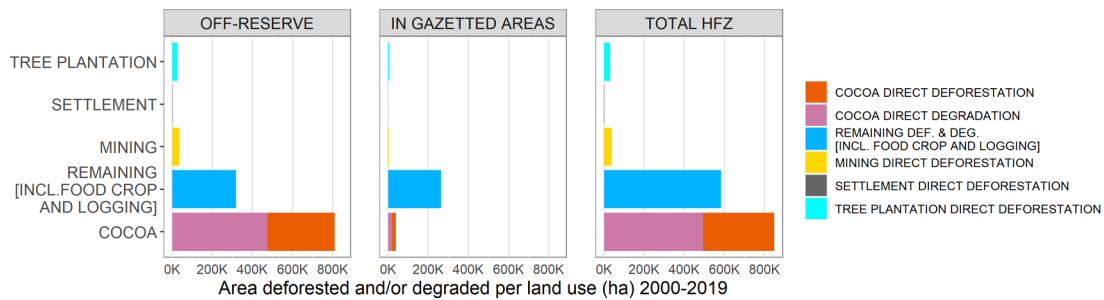


Figure 3.5. Deforestation and degradation per driver off-reserve, in gazetted areas and in the entire high forest zone over 2000-2019. Gazetted areas include forest reserves, national parks and game reserves. ‘Remaining’ refers to the deforestation and degradation not attributed to any satellite-based driver (thus including food crops and logging).

3.3.3. Scale-dependent patterns

The above calculations were made at the region-level before being summed, capturing the local dynamics in each region. This approach thereby assumed that there were no land-use dynamics across regions, i.e., each region functions in isolation. When repeating the same procedure directly at the level of the whole HFZ, the contraction and expansion dynamics occurring in distant locations average out and hence affect the overall outcome. Over 2000-2019 in the HFZ, cocoa is the land use which expanded the most, with a net expansion area of 790,000 ha (582-961 Kha), from 1.87 Mha (1.7-2M ha) in 2000 (which, when removing the 24% unproductive cocoa farmland, is similar to the 1.3 Mha of harvested area reported by FAO for 1999-2001) to 2.66 Mha in 2019. Net expansion of food crops was 500,000 ha, mining 113-141,000 ha, tree plantations (including rubber, palm oil and coconut) 54-104,000 ha, and settlements 70,000 ha (SI Figure S5).

Yet, at the HFZ level, the net expansion area of cocoa on land not forested in 2000 was negative over that period (-57,000 ha, -266 Kha/+113 Kha). Cocoa thus expanded massively into forests, replacing 848 Kha of forest, but on net, contracted or barely expanded outside forests, essentially in the Eastern and the Ashanti Regions. This reflects the historical shift of the cocoa frontier from eastern to western Ghana, in search for available land and fertile soils, while the old unproductive cocoa farms were slowly abandoned and converted to other land uses such as oil palm or food crops (Asubonteng et al., 2018; Ruf, 1987). In the Eastern Region, cocoa replaced 108,000 ha of forests but shrunk outside forests (-265,000 ha (-314/-224 Kha), with a net contraction of -156,000 ha (-205/-116 Kha). Similarly, cocoa moved into forests in the Ashanti Region, with 224,000 ha of forest converted to cocoa, but lost an area of 87,000 ha (-142/-41 Kha) outside forests, resulting in a net expansion of 137,000 ha (80K-183Kha). In contrast, the other four

regions experienced an expansion of cocoa both into and outside forests, particularly in the Western Region (which here includes Western North), where cocoa expanded by 427,000 ha (354-488 Kha) (62% of the total land use expansion in the region), of which 121,000 ha outside forests.

Cocoa expansion outside forests in the west-ward regions is smaller than contraction in Eastern and Ashanti Regions. As our methods use the expansion of cocoa outside forests to attribute indirect deforestation, the HFZ-level analysis attributes limited (0-7%) food crop deforestation and degradation to cocoa.

3.3.4. Projecting trends in the future

Cocoa is the dominant land use in the HFZ of Ghana in 2019, occupying 34% of the entire area, and nearly half of the Western Region (Figure 3.6A). This region, in addition, has no remaining pool of land other than food crops and forest into which cocoa or other commodity land uses could potentially expand, indicating a saturation of the landscape. As a simple thought exercise, we projected the past 2000-2019 annual expansion rates of each land use to the next 2020-2030 decade, and compared this to the remaining forest and other land area. In this exercise, the potential expansion, mainly of cocoa, would necessarily have to occur at the expense of the remaining undisturbed tropical moist forest in the Western Region (Figure 3.6B). In the Central Region, the projected expansion of land uses would exhaust almost all the remaining forest area. In the Eastern Region, the contraction of cocoa area over the last two decades might release land for other land uses. Even when some ‘other’ land is available for expansion, converting them for food crops, cocoa or other commodity crops would entail trade-offs, as this land might include fallows that are part of the agricultural system, bushes and other natural vegetation providing various products and services.

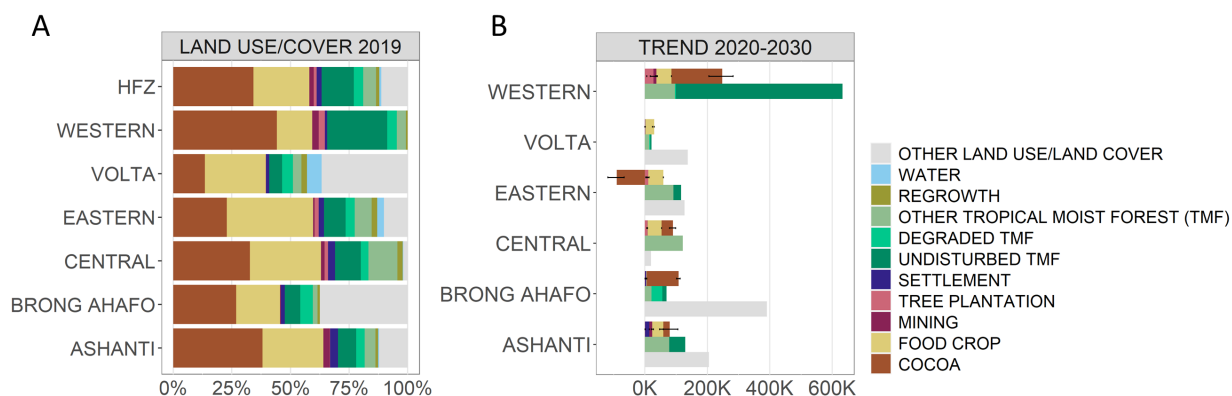


Figure 3.6. Land use in 2019 and projection over 2020-2030. (A) Percentage of each region (restricted to HFZ) occupied by each land use in 2019-2020. ‘Other tropical moist forest’ refers to trees lower than

8m in patches smaller than 1ha patches in 2019 (from which cocoa was masked out). ‘Other land uses/land covers’ refers to land not occupied by the other considered land uses – which could correspond to bush, fallows, grasslands, or cashew and orchards not included in our analysis. **(B)** Trend over 2020-2030 of the expansion area of each land use annualised over 2000-2019 and multiplied as a rough projection over 2020-2030 (above stacked bar), compared to the remaining tropical moist forest (TMF) in 2019 (middle stacked bar), and to the ‘other land uses/covers’ remaining in 2019 (bottom bar).

3.4. Discussion

3.4.1. Direct and indirect deforestation and degradation: comparison with other studies and limitations

Comparison with other studies

Our results show the large footprint of cocoa on Ghanaian forests, making it the major direct, but also indirect, deforestation driver over 2000-2019, ahead of food crops, logging, mining, tree plantations and settlement sprawl combined. Food crops and logging are the next two major deforestation drivers according to our analysis. Masolele et al. (2024), while not accounting for indirect effects, similarly find that mining, tree plantations, and settlements were minor drivers of deforestation in Ghana, though their analysis at the national scale highlights the relative importance of food crops versus cocoa when including the transition zone and savannah regions (occupying 37% vs 25% of land deforested from 2000-2019, respectively). Ghana’s REDD+ report (Forestry Commission, 2016) attributes 35% of the national deforestation and degradation to wood harvesting (vs. ~16% from our findings for the high forest zone), 10% to population and development pressures (vs. <1%, knowing that the HFZ excludes the Greater Accra Region), 5% to mining and mineral exploitation (vs. 3%) and 50% to agricultural expansion, versus 83% from our findings when accounting for all agricultural post-clearing land uses (i.e. cocoa, food crops and tree crops). Our estimate is close to Acheampong et al. (2019)’s finding of 78% of the forest loss caused by the expansion of annual crop farms and tree crops in the Ashanti Region. Using Hansen et al. (2013) data as a foundational layer to define natural forests (vs. JRC TMF), Singh and Persson (2024) find ~437,000 ha of cocoa deforestation over 2001-2019; 83,000 higher than our cocoa deforestation estimate (excluding degradation) which also restricts forests in 2000 to >8 m canopy height and >1 ha patches. Sub-regional landscape-level studies find a contribution of cocoa to forest loss between 24% in the Eastern Region (Asubonteng et al., 2018) (vs. 53% from our findings, SI table S7), and >50% in the Western Region (Ajagun et al., 2022; Ashiagbor et al., 2022; Benefoh et al., 2018) (SI table S1) (vs. 59%).

According to our analysis, mining was not a major deforestation driver between 2000 and 2019. The mining dataset used in our research (Maus et al., 2022) detects 140,000 ha of large-scale and artisanal and small-scale mining sites in the study area in 2019 (191 Kha in the entire Ghana). Although the mining area has increased rapidly in Ghana over recent years (doubling from 31 to 67 Kha between 2019 and 2024 for artisanal and small-scale gold mining only, according to CERSGIS (2025)), the land footprint of mining remains limited compared to other land uses, especially cocoa (2.66 Mha) and food crops (1.86 Mha). Importantly, the environmental impacts of mining activities go well beyond deforestation and are significantly harder to reverse or restore, reducing the possibility of using the land for other purposes in the future (Boateng et al., 2014; Eberhard et al., 2022; Obodai et al., 2024) (see text S2 for further details on mining).

Complex land use transitions

Land use change dynamics are complex and deforestation or forest degradation rarely have a single underlying cause (Pendrill et al., 2022a). In a context of unstructured, collective harm, building on Brülde et al. (2023), and in the absence of annual time series of land use data, we attribute the impacts to land uses based on “participation”, here assessed as the contribution to additional land demand in the region, rather than counterfactual reasoning. We assessed deforestation drivers using the *net* land use expansion, thus not capturing gross transitions between land uses. It is possible that other land uses occurred between the deforestation event and the land use observed in 2019. Yet in the context of the 2000s’ cocoa boom and the comparative perceived advantages of cocoa, there is convincing evidence that land detected as cocoa in 2019 was initially deforested with the intention of growing cocoa (Ashiagbor et al., 2022; Benefoh et al., 2018; Clough et al., 2009; Ruf, 2007a). Besides, our research was limited to the indirect deforestation linked to the displacement of food crops only, and thus does not reflect all indirect dynamics at stake. Land uses like mining can also have other indirect effects such as attracting workers who will settle nearby and grow their own crops – thus increasing mining’s indirect role in driving deforestation (Giljum et al., 2022). The land balance approach used here to identify indirect drivers of deforestation through food crop displacement captures how different land use expansions contribute to the overall pressure on land in a landscape but does not disentangle or trace these complex land use trajectories and underlying causes. Although these limitations restrict our ability to precisely assess causal land-use dynamics, our analysis provides the most convincing evidence so far on these dynamics.

Entanglement of logging with other drivers

In addition, limitations inherent to spatial and statistical datasets may produce uncertainties in the results, though there is no evidence to suggest specific biases in one or another direction (table S2). The role of logging (legal and illegal) and wild-fires in deforestation and degradation (~16%, versus 35% in Ghana's REDD+ report) is particularly hard to disentangle from food crops and other land uses. They are highly intertwined, with for instance farmers felling trees to get enough light to start their crop, or establishing their crop on land originally cleared for timber. Within forest reserves, the 'Modified Taungya System' (MTS) allows specific community groups to temporarily farm degraded land for food crops, provided they plant tree seedlings within their farm and stop farming once the canopy closes. Though we could not access MTS maps to assess the proportion it represents, logging's prevalence in deforestation is certainly higher than we estimate. Logging could thus be involved in between ~16% (our conservative estimate, which prioritises attribution to food crops) and up to ~40% of the overall deforestation and degradation, if considering that all food crops in forests were established on already degraded land – or even higher, knowing that cocoa expansion also tends to follow the relics of previously logged areas (Benefoh et al., 2018).

Land balance model uncertainties

Similarly, our results represent a conservative – and probably underestimated – quantification of indirect deforestation, especially for cocoa. No preferential criteria were applied to the expansion of land uses; they had the same probability to expand on any of the available pools of land. However, as no quantified estimate of these land use trajectories could be made, we maintained the neutral assumption as a conservative approach. Also, estimates of the land use areas in 2000 (and thus their expansion area) contain uncertainties (table S4, text S1.4). As an example, absent sub-national estimates, we added the same 24% of unproductive cocoa area in 2000 to each region, which may overestimate the initial cocoa area in regions having more productive farms. Consequently, the cocoa expansion after 2000 is probably underestimated, leading to an underestimation of the share of indirect deforestation attributed to cocoa. Further, in our land balance model, regions at the periphery of the cocoa-growing landscape, such as Brong Ahafo, Central, Eastern and Volta Regions, include more 'other' land (Figure 3.4A) which absorbs a substantial part of the land use expansion and food crop displacement. This reduces the share of food crop and indirect deforestation and degradation, and may thus not capture sub-regional realities. Finally, additional uncertainties arise from the fact that pressure on food crop area could induce a contraction (through intensification, or more food imports and/or worsen food security) or an increase in food

crop area (if displaced on less productive land). In the absence of evidence suggesting one or the other, our model displaces one hectare of food crop for each converted hectare.

Scale-dependent patterns

In attributing deforestation to land uses, the scale of analysis matters, especially for land-balance models (zu Ermgassen et al., 2024). This scale dependency can explain the difference between our results and those of Pendrill (2022b), who use a similar land-balance model for attributing deforestation but at national-level, based on FAO agricultural statistics. At national scale, local ecoregions and land use trajectories are not accounted for, and cocoa is thus considered as expanding on pastures (though these are mostly located in the savannah regions where cocoa does not grow) rather than forests – resulting in no deforestation attributed to cocoa in Ghana according to their study (SI table S1). Using a land balance approach, the role of cocoa thus appears when focusing on the regional dynamics within the high forest zone. Similarly, the role of mining appears very important but at a more local scale, such as within mining concessions, where mining was found to be the major driver of deforestation and to displace farmland within forests (Schueler et al., 2011).

3.4.2. The role of cocoa expansion in driving food crop deforestation

Over 2000-2019, most deforestation and degradation happened off-reserve, where cocoa has been the lead driver and is legal (Ashiagbor et al., 2022). Our results indicate that cocoa directly drove little deforestation in forest reserves, where food crops and logging were the major direct drivers of forest loss. Yet, as observed by several studies (Ajagun et al., 2022; Kumeh et al., 2022) and as shown by our land balance, though direct cocoa-driven deforestation in reserves is relatively limited, cocoa expansion indirectly plays a large role by displacing food crops into forests – mainly because the massive expansion of cocoa has led to a saturation of the landscape, especially in the Western Region, with cocoa occupying 44% of the region resulting in ‘monotypic cocoa landscapes’ (Ajagun et al., 2022) (Figure 3.6).

This relates to well-described cocoa cycles (Clough et al., 2009; Ruf & Siswoputranto, 1995). In a new cocoa basin, farmers take advantage of the ‘forest rent’, providing ‘vacant’ land and high yields from nutrient-rich soils generating good income, thus attracting more labour and creating a production boom (Acheampong et al., 2019; Benefoh et al., 2018). After ~20 years, cocoa farms

become less productive due to ageing farmers and trees, depleted soils and the increasing prevalence of pests and diseases. In parallel, the production surplus may have led to decreasing market prices when the affected area is large enough. The drop in production, combined with limited income for farm rehabilitation, eventually leads to a cocoa bust. As a result, market prices may rise again, favouring the start of a new cycle – with the opening of a new forest frontier and the emergence of a new cocoa production basin where forest and labour is available. This boom was observed for instance in the Western Region in the early 2000's (Ruf, 2007a). The start of the cycle, involving many young cocoa farms, is coupled with high food production (intercropped with cocoa seedlings), while the end of the cycle is characterised by land saturation, which, coupled with population increase, makes available land off-reserve scarce (Ashiagbor et al., 2022; Benefoh et al., 2018; CFI, 2023; Kumeh et al., 2021; Kyere-Boateng & Marek, 2021). When the land balance model is applied at the high forest zone level, the contraction of cocoa in the Eastern Region cancels out the displacement of food crops by cocoa in the western part. This reflects the east-west shift of cocoa in Ghana and the fact that, overall, the *net* cocoa area expansion across Ghana was smaller than the *gross* expansion in the western part of the country, but this hardly reflects the socio-ecological impacts of the dynamics at play. Displacement of land uses can happen within a locally-saturated area, with farmers forced to clear new patches of land – legally or illegally – to plant their food crops, even if land would be available in a distant location.

Local land saturation by cocoa can be considered as the main mechanism driving food crop deforestation, via two dynamics. First, cocoa replaces food crops off-reserves (Ajagun et al., 2022; Asubonteng et al., 2018, KI4,7). Farmers make this shift because they lack land and are searching for financial security, which can be better ensured by cocoa than food crops (as cocoa is paid in bulk, with a fixed price, and can pay for school fees for instance) (Ajagun et al., 2022). To a lower extent, converting food crops to cocoa is also linked to societal norms (being a cocoa farmer provides a higher social status) and to the prevention of land tenure conflict – cocoa being a way of securing land ownership, at least for the lifetime of the cocoa trees (Ajagun et al., 2022; Hirons et al., 2018). Second, even if no further cocoa expansion would occur, by occupying a vast share of the landscape, cocoa leaves little land available for new food crops, as shown above. Farmers can sometimes find no other option than to look for land in forest reserves (Ashiagbor et al., 2022; Benefoh et al., 2018; Kumeh et al., 2021, 2022). Growing food crops within forest reserves is perceived as less risky than growing cocoa, which requires more time to become productive and more investments – while being less tolerated

by forestry officials. Thus, to avoid seeing their efforts cut down by forest guards, farmers tend to favour food crops over cocoa in forest reserves (KI2-4,6, FG1). This differential perception of risk also reinforces the conversion of food crops to cocoa; anticipating that growing food crops on-reserve is, at least to a certain extent, possible, farmers in communities fringing forest reserves may count on forest reserves as a pool of land for their food crops. This pattern may however vary among communities; e.g. cocoa happens to be illegally grown in forest reserves, particularly where admitted farms are located (Benefoh et al., 2018; Brobbey et al., 2020; Kumeh et al., 2021).

The lack of land, the conversion of food crops to cocoa and their subsequent displacement in forests can be seen as ‘push’ factors – pushing food crops in forests. The Modified Taungya System on the other hand acts as a ‘pull’ factor, facilitating the entry of food crops in degraded forest reserves – legally at first, but in many instances making way for further encroachment beyond the initial delineations (Ankomah et al., 2020; Kumeh et al., 2021). Other factors play a role, such as forest soils’ fertility (Ruf & Siswoputranto, 1995), weak law enforcement (Acheampong et al., 2019), the ‘right to food’ and rights to the land claimed by some communities (Acheampong et al., 2019; Kumeh et al., 2021), and poverty and the lack of alternatives (Ashiagbor et al., 2022). Land use segregation is thus underway in communities fringing forest reserves, with off-reserve land increasingly dedicated to cocoa and other cash crops, and on-reserve land mainly used for food crops, either legally or illegally.

This may explain the seemingly contradictory discourses around deforestation in Ghana. Amidst the debates aiming at attributing the responsibility of the rise in deforestation and degradation between 2017 and 2018 (+60%) to specific drivers, Dekker (2019) pointed at large scale non-cocoa agricultural expansion (thus including food crops), logging, mining, and fire, while Weisse and Goldman (2019) emphasised the role of cocoa expansion. Reconciling these discourses requires to note that, although – as shown by our results – cocoa was historically by far the main direct driver of deforestation and degradation in Ghana, in recent years, forest has shrunk up to the point of being limited to forest reserves where cocoa is more strictly forbidden and thus more difficult to cultivate, making logging and food crop production the new main observed direct drivers of deforestation and degradation – as reflected by the assessment of the CFI priority landscapes identifying food crops as the main deforestation driver in 2019-2022 (CFI, 2023). Yet, behind the scenes, cocoa continues to play a determining role in land use dynamics and deforestation.

3.4.3. Is deforestation-free cocoa possible?

Deforestation leakage

Indirect land use change, as shown by our results, can represent an important deforestation leakage for policies aiming at halting deforestation, such as private zero-deforestation commitments (ZDC) (Lambin et al., 2018) or the European deforestation-free regulation (EUDR). First, as long as such policies remain individual company choices (i.e. ZDCs), or are enforced only for a single export-market (i.e. EUDR), the whole sector is not covered and a ‘supply chain leakage’ can occur, where farmers continue to produce cocoa linked to deforestation but shift to other buyers (also referred to as “market segmentation”) or sell their products by “laundering” them through intermediaries (Meyfroidt et al., 2018). As an example, the state-owned Produce Buying Company – second largest licenced buyer in Ghana – has not adopted a zero-deforestation commitment, while Asia is the second largest destination for cocoa beans from Ghana (25%), offering a way out for cocoa that does not comply with ZDCs or the EUDR (COCOBOD, 2020). Companies may favour sourcing from low-risk regions – away from remaining forests; however, non-compliant cocoa will still continue being traded by uncommitted intermediaries and/or will find its way to the non-EU market (Bakhtary et al., 2020). Second, policies such as ZDCs or forest protection (e.g. forest reserves) limit the availability of land for further expansion of cocoa but lead, as shown by our results, to indirect land use change – with cocoa expanding on food crops and saturating space off-reserve and food crops being displaced in forests. This could be identified as an ‘activity leakage’, where the land use targeted by restrictions (cocoa) concentrates on allowed lands (off-reserve), and labour and capital used on these lands are reallocated to places with available and accessible land (here food crop farming being reallocated to forests) (Meyfroidt et al., 2018). Though our results do not demonstrate the causal attribution of this displacement to ZDCs and forest protection interventions, the forthcoming EUDR may further strengthen this spill-over mechanism by increasing the surveillance of cocoa deforestation.

In practice, these supply chain-focussed policies have had only limited success (Garrett et al., 2018) and, because they do not account for deforestation leakage, their potential conservation outcome will continue to be curtailed (Gardner et al., 2019; Garrett et al., 2019; Gollnow et al., 2022; Lambin et al., 2018). Increasingly, landscape or jurisdictional approaches, where stakeholders collaborate in a specific landscape or jurisdiction to promote sustainable resource use (Reed et al., 2016; von Essen & Lambin, 2021), are put forward as a complementary tool in addressing deforestation. They indeed have the potential to internalise indirect sourcing and deforestation drivers beyond the focal commodity (Parra Paitan, 2024; Parra-

Paitan & Verburg, 2022; zu Ermgassen et al., 2022). However, it is unlikely that these initiatives would rapidly cover all forests across the tropics, thus keeping the door open to deforestation leakage. Landscape approaches are thus highly relevant and necessary to preserve forests while ensuring a just transition for involved stakeholders, but require complementary tools to halt deforestation – including a way of addressing key underlying drivers of deforestation.

A more transformative vision: Addressing key underlying drivers of deforestation, industry growth, and economic dependency

Ultimately, the saturation of the landscape by cocoa and thus the direct and indirect deforestation in Ghana and other similar cocoa-dominated regions is rooted in a growing cocoa sector, driven by an ever-growing international cocoa industry and a global demand increasing at ~3% per year (Beg et al., 2017), requiring more land. Yields could certainly be improved in order to slow the need for expansion, yet, with cocoa yields stagnating around ~500 kg/ha in Ghana and Côte d'Ivoire for the past decade despite industry's efforts (Asante et al., 2021), and the increasing adverse impacts of climate change (Schroth et al., 2016), it is unlikely that cocoa yields could be raised enough to decouple the increasing international demand from land area used – as a comparison, the highest increase in yields during the Green Revolution was 5% per year for wheat (FAO, 2004). Further, an increase in yield may generate a rebound effect, spurring even more forest conversion to cocoa (Byerlee et al., 2014; Goulart et al., 2023; Rodriguez García et al., 2020). Though largely disregarded by deforestation-free policies (Kumeh & Ramcilovic-Suominen, 2023), the overall level of consumption of cocoa and other deforestation-risk commodities by high-income countries should be part of the discussion to reach the goal of deforestation-free landscapes. Shifting from mass consumption of low-priced bulk cocoa to smaller consumption of well-paid cocoa could limit negative impacts on cocoa households and producing regions.

Nevertheless, beyond deforestation, whether or not Ghana (or Côte d'Ivoire, as another major cocoa producer) should keep focusing on cocoa as a key economic sector – leading to land saturation for a single commodity dedicated to export – can be questioned. Specialising in export-oriented cash crops generates multi-faceted dependencies to international market demand, energy prices (e.g. for transport, transformation, synthetic inputs), and domestic production capacities – all vulnerable to increasing climate shocks and, in the mid-run, to decreasing affordable fossil fuel resources (Daruich et al., 2016; de Roest et al., 2018; UNCTAD, 2023). By occupying land, export-crops directly compete with food crops destined for the domestic market, and can put food availability and access at risk in case of disruptions in productivity or low international market prices of

export crops relative to food crops (Anderman et al., 2014; Feyaerts et al., 2020). Recent signs of the emergence of a structural cocoa recession in Ghana and Côte d'Ivoire, following the cocoa cycles (Ruf, 2024), highlight the risk of a strong dependency on cocoa. If planned with appropriate policies and political will, down-sizing the importance of cocoa in the economy could free up land for diversification from export-dependent crops and participate in building resilience. Some farmers are currently making this move away from cocoa by establishing permanent food crops (Hirons et al., 2018).

Yet, a progressive diversification away from the single dominance of cocoa (or other export-crops) in major producer countries would require lifting structural 'lock-ins' such as, at the country level, the dependency on earning foreign currencies – required notably to service financial debts (World Bank, 2023), as well as the vested interests in cash-crop production and trade from actors in government, companies and some civil society organisations (Odijie, 2018). At the farmer level, the 'cocoa poverty trap' acts as a major lock-in – where farmers are unable to gather the required capital to rehabilitate their farm or shift to another crop due to a combination of factors, including low cocoa income from low value capture by farmers in the cocoa supply chain (BASIC & FAO, 2020) and low yields of ageing farms (Hütz-Adams, 2022; van Vliet et al., 2021).

Hence, policies addressing poverty to lift these lock-ins must also be part of the discussion – including a structural transformation in international trade relations, rebalancing power and wealth distribution (Hickel et al., 2022). Scientists, indigenous people, policy makers and civil society around the world are now increasingly advocating for more transformative visions such as doughnut economies (Raworth, 2017), foundational economies (Bentham et al., 2013), socio-bioeconomies (Garrett et al., 2024), democratic economic planning (e.g. Devine, 1988) or degrowth (Weiss & Cattaneo, 2017), among others. These calls include ideas that challenge continuous growth, support structural transformations of the economies away from extractive and degrading processes towards more regenerative and restorative approaches, and recenter policy objectives around sustainability, justice, diversity, and resilience. Compared to current isolated deforestation efforts, such approaches offer new avenues to address the lock-ins and complexities currently affecting the West African cocoa sector to tackle deforestation in a more systemic way.

3.5. Conclusion

We found that cocoa was the main direct but also indirect driver of deforestation and forest degradation in the high forest zone of Ghana between 2000 and 2019,

followed by food crops and logging. Mining, tree plantations and settlements had, comparatively, a limited direct impact on forests over this period, though rapid mining expansion has raised attention over the recent years. Cocoa mainly replaced off-reserve forests, with a relatively limited impact in forest reserves where food crops and logging, as well as other factors such as wildfires, were the major direct drivers of deforestation and degradation. Our land balance approach shows that about a fourth to a third of the forest land converted to food crops corresponded to food crops being displaced by other land uses – with half of this displacement being due to cocoa expansion. In addition to the conversion of food crops to cocoa, the saturation of land by cocoa, especially in the Western and Western North Regions, forces farmers to search for land within forest reserves which they mainly use to grow food crops, either legally, under the Modified Taungya System, or illegally. Accounting for both its direct and indirect impacts, cocoa is an important pressure on these gazetted forests. These findings show the need for private zero-deforestation commitments and public legislations to move away from narrow, single commodity-focused initiatives aimed at halting deforestation. Instead, these initiatives should take into account the dynamics of the entire landscape, acknowledge the systemic linkages between cocoa, food crops and other land uses, integrate all drivers and anticipate rebound and leakages. Most importantly, underlying causes should be tackled, such as the industry growth, calling for demand-side actions, and poverty, demanding structural changes in international trade relations, rebalancing power and wealth distribution.

3.6. References

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Chapter 4

Global climate and biodiversity targets on a limited and used planet

This chapter is in preparation for submission.

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Abstract

To mitigate the ongoing ecological and climate crisis, governments and private actors are developing multiple interventions, among which three have large implications on land: (i) conservation of natural ecosystems, (ii) forest and landscapes restoration and afforestation and (iii) non-fossil energy production, through renewable energy infrastructures and the associated minerals mining, and land-based bioenergy. Expanding the area devoted to these sustainability interventions can lead to conflicts between these and with other land uses. We compiled quantitative targets from multilateral agreements or public commitments, and, using global suitability and priority maps, provide a spatial assessment of their distribution and overlap at the global level. We found that meeting global sustainability targets would require more than doubling the area already occupied by the considered interventions, to reach a total of ~5350 Mha in 2050, or ~40% of the ice-free land surface (of which 30% for conservation). A third of current agricultural land, and over half of the forestry land would be targeted in 2050 if the interventions were spatialized following a top-down and siloed approach, with large country and regional disparities. Over a third of energy-related best-locations are also targeted by other interventions. When also considering other business-as-usual land demands for food, fiber and timber, taken together with sustainability interventions, the area needed is larger than Earth's ice-free land surface. Our findings provide additional insight into the mounting land squeeze, and stress the need to break silos, and focus on organising sufficiency to reduce production and overconsumption.

4.1. Introduction

Governments and private companies are increasingly making pledges aiming at mitigating the ongoing ecological and climate crisis, under the umbrella of three major UN conventions' targets: the CBD's Kunming-Montreal Global Biodiversity Framework (GBF) aiming among others at restoring at least 30% of degraded ecosystems and at conserving at least 30% of terrestrial and marine areas by 2030 (CBD, 2022); UNCCD land degradation neutrality (LDN) aiming for no net loss of healthy and productive land by 2030 (UNCCD, 2015); and the UNFCCC Paris Agreement to hold global warming well-below 2°C and pursue efforts to limit it to 1.5°C (UNFCCC, 2015). Three main groups of interventions expected to play a major role therein have large implications on land demand: (i) the conservation of natural ecosystems, (ii) restoration of degraded terrestrial ecosystems and afforestation for biodiversity and/or carbon benefits (shortened to 'restoration' hereafter) and (iii) the development of renewable energy infrastructures, with their related mining footprint, and the expansion of land-based bioenergy.

A body of research argues that halting the dual crises of ecological breakdown and climate change requires protecting large shares of Earth's land and oceans (Allan et al., 2022; Dinerstein et al., 2020; Wilson, 2016). The UNCCD's Glasgow Declaration (UNCCD, 2021) and UNFCCC's Global Stocktake (UNFCCC, 2024) committed to working towards halting and reversing deforestation and forest and land degradation by 2030. Then, beyond the GBF, a growing momentum drives commitments for ecosystem restoration and afforestation for carbon sequestration via multiple initiatives such as the Bonn Challenge, AFR100 or the Trillion Trees Campaign (van der Esch, 2022). Further, most climate stabilisation scenarios rely on a steep rise in renewable electricity production by 2050, mainly solar and wind power (IEA, 2024). These technologies have substantially lower energy densities than fossil fuels (Fritsche et al., 2017; van Zalk & Behrens, 2018), and imply an expansion of mining activities (Luderer et al., 2019; Watari et al., 2019), leading to growing concerns about their land use impacts (Sonter et al., 2020; Stevens et al., 2017). Finally, despite limited effectiveness in mitigating climate change, and potential adverse effects (Creutzig et al., 2021), bioenergy is included in many scenarios to stay within the Paris Agreement, as fossil fuel substitute and, when combined with Carbon Capture and Storage (BECCS), as a way to absorb carbon (IPCC, 2022a).

On an already used planet (Meyfroidt et al., 2022), devoting more area to these interventions might lead to conflicts among these land uses, as well as with other

land uses also serving crucial societal goals such as agriculture for food production, and forestry. This competition has been framed as ‘land squeeze’ (Erb et al., 2024; IPES-Food, 2024; Searchinger et al., 2023), ‘land crunch’ (King et al., 2023), ‘land gap’ (Dooley et al., 2022), or ‘land scarcity’ (Lambin & Meyfroidt, 2011). Expansion of some land uses can displace others and generate indirect land use change (Meyfroidt et al., 2020), and can increase pressure on ecosystems, food security and local communities’ rights (Osborne et al., 2021; Sovacool et al., 2020; Woods, 2019). Conversely, some of these land uses could coexist or even foster synergies (Sayer et al., 2013).

Sustainability initiatives tend to work in silos (Lah, 2025; Pörtner et al., 2021), lacking coordination and systemic approaches, hence limiting their ability to prevent potential adverse rippling effects (Meyfroidt et al., 2022; Pascual et al., 2017). Global-scale assessments of the potential and spatial prioritization of land-based interventions are typically made separately, depicting opportunities for a single or a limited subset of interventions; e.g., for re/afforestation (Bastin et al., 2019), restoration (Griscom et al., 2017; Strassburg et al., 2020), conservation (Dinerstein et al., 2020; Jung et al., 2021; Mo et al., 2023), biomass production (Li et al., 2020; Naess et al., 2021; Oakleaf et al., 2019), or onshore wind and utility scale-solar (Oakleaf et al., 2019; van de Ven et al., 2021). Some efforts encompass several dimensions such as carbon, biodiversity and water (Jung et al., 2021), but few studies integrate more than a subset of interventions (Duarte et al., 2023; Fuhrman et al., 2023; Neugarten et al., 2024). Several studies looked at how different interventions may compete, but are not thematically or geographically comprehensive; e.g., bioenergy crops vs. solar power (Sturchio et al., 2025; van de Ven et al., 2021), vs. afforestation (Gvein et al., 2023), or vs. biodiversity (Prudhomme et al., 2020), conservation vs. afforestation (Hong et al., 2020; Seddon et al., 2021), or overlaps of suitability maps for land-based climate mitigation measures (Beaury et al., 2024). A few other studies account for social impacts, but typically focusing on one intervention (Fesenmyer et al., 2025). By perpetuating siloed approaches, these mapping efforts risk promoting unjust outcomes (Obura et al., 2021; Windey, 2020). More holistic assessments of the multiple land uses and land demands are required to accurately represent realities of implementing pledges (Benra et al., 2024). Along this line, integrated assessment models aim to identify scenarios that reconcile these different demands (e.g., Leclère et al., 2020; Venier-Cambron et al., 2023; Weindl et al., 2024) but (i) work under a lot of assumptions on e.g. economic optimization and prioritization of solutions, and (ii) by deriving solutions, they do not highlight tension points and possible overlaps that might arise from assessments made separately across interventions. So far, no study has

comprehensively and empirically assessed the land demand and plausible geographic prioritization of all the major land-based sustainability interventions.

Here, we aimed to assess, globally, to what extent major land-demanding sustainability interventions might be competing for the same land by 2050 and would be driving additional pressure on existing land uses. Specifically, we asked (i) how much land is required to meet major global sustainability targets, and how does this add up to the business-as-usual (BAU) other land demands? (ii) Where are the areas identified as being the most suitable for these interventions, and which could thus be targeted? (iii) What is the current land use in these areas? (iv) How much and where could these interventions overlap, by targeting the same locations? and (v) How are countries differently impacted by these interventions and their possible overlaps?

First, we compiled quantitative targets per country of the major land-based interventions pushed by the global sustainability agenda, as expressed through multilateral agreements or public commitments (see Methods). Second, we spatialized these targets using global suitability or priority maps from recent works to identify the plausible distribution and overlap of these interventions, if rolled-out in an uncoordinated, top-down and suitability-driven way, at the global level by 2050. To do this, we selected the considered best-suited locations for each target, per country. We derived global, binary, 1x1 km resolution maps of the distribution of each intervention's targeted location in 2050. By stacking these, we identified areas of overlap between considered interventions, as well as with major existing land uses – and how they vary across countries. We also assessed recent estimates of land demands for built-up areas, pasture, cropland and forestry for timber for the coming decades, retaining only scientifically-robust estimates of business-as-usual situations, excluding disruptions, transformations or enhanced policy efforts (table S1).

We made several assumptions and methodological choices:

- 1) We focus on formally made commitments under the CBD and UNFCCC, and additional quantifications of the plausible implications that derive from these commitments (see assumptions 5 and 7). We did not integrate private companies' commitments or pledges to avoid double counting, as governments' accountings integrate both public and private projects. We did not include interventions aiming at improving the sustainability of land management itself without land use conversion (e.g. cropland transitioning to agroecology).

2) We spatialized each intervention separately before overlaying them, thus assuming independent planning mechanisms.

3) For conservation, absent any explicit burden-sharing in the GBF itself, we considered that every country implements its responsibilities to conserve 30% of the most valuable land for biodiversity within its own territory and for every biome, without offshoring to other countries.

4) For restoration, we used governments' pledges to the UNFCCC compiled by the Land Gap report (Dooley et al., 2022) and implying land use change, i.e., not including restoration of degraded ecosystems that maintain the ecosystem type (e.g., restoring degraded forest to healthier forest). Our estimate thus mainly consists of reforestation and afforestation for climate change mitigation, hereafter referred to as 'carbon-oriented restoration'. We assumed that these pledges also contribute to the CBD Target 2 of 30% restoration of degraded ecosystems, although noting the criticisms of considering carbon-oriented afforestation as a form of restoration. For countries without publicly-available pledge, we used the average country area percentage pledged by the other countries for restoration involving land use change.

5) As a plausible implication of countries' commitments under the UNFCCC's Paris Agreement, we used a 100% Wind, Water, and Solar (WWS) renewable energy scenario for 2050 allocated by country (Jacobson et al., 2022), adding the land use demand for mining energy transition minerals (ETMs) (Owen et al., 2022).

6) For wind power and ETM mining, we calculated the area plausibly impacted by the intervention, i.e., 1 km² grid cells with windmills and 10 km buffers around mining sites, not the direct physical footprint of the windmills or mining sites.

7) For bioenergy (BE), we used national pledges compiled by the Land Gap report for five countries, complemented for other countries by the Net Zero 2050 roadmap of the International Energy Agency (IEA) (IEA, 2021). This scenario is presented as the lowest land demanding, with bioenergy essentially serving for residual demand in hard-to-electrify sectors not covered by the 100% WWS scenario.

Our findings inform on potential conflicts and challenges in implementing land-demanding interventions considered within the global sustainability agenda. This assessment is not a forecast of what will really happen in 2050, or a normative scenario, but an overview of the global land demand for the coming decades, an

exploration of what land might be targeted under absence of coordination across interventions, and a call to rethink the role of land in addressing the climate and ecological breakdown.

4.2. Results

4.2.1. Total land demand for sustainability interventions

In our best estimate, summing up the entire land requirement for each intervention in 2050, 5347 Mha (5200 – 6148 Mha) would be required, i.e. 41% of the ~13,100 Mha ice-free land surface (Tab. 1, Figure 4.1A), on a planet where human societies already use or manage ~75% of the land surface (IPCC, 2022b; Meyfroidt et al., 2022). Currently, these interventions occupy about 2454 Mha. More than doubling this, an additional 2888 Mha would thus be required by 2050 to reach the considered pledges and targets.

Most of that area is for conservation, yet, this vision for 2050 involves massive land use changes, as ~8.9% of the planet would be needed for renewable electricity, bioenergy, mining for ETMs, and restoring used land to natural ecosystems, +8.1% compared to today. This compares, for example, to current cropland covering ~12% of the planet's land. Renewable electricity and ETM mining would cover ~2.2% of the global land – with ~1.1% of potentially highly artificialised surfaces -, around twice the current global extent of built-up land. Our interventions amount to an additional annual land demand of ~96 Mha/y in total, and ~19.8 Mha/y when excluding conservation and restoration. This compares to net rates of cropland expansion of ~4 Mha/y over the recent decades (FAO, 2024), net forest loss of ~-5 Mha/y (FAO, 2020), or ~6.4-8.8 Mha/y of agriculture-driven tropical deforestation (Pendrill et al., 2022).

The +2888 Mha land use demand for climate and biodiversity interventions, +1060 Mha when excluding conservation, compares to a BAU projected increase in land use demand of +510 Mha for agriculture, built-up land, and forestry for timber (with a large range of 125-888 Mha, due mainly to uncertainties on grazing), highlighting the large role that these interventions are likely to play in future land pressure and competition. Together, all land demands for 2050 correspond to ~14,244 Mha, to fit into an Earth's land area of 13,100 Mha including snow-covered land and deserts.

Table 4.1. Estimates of current and additional land use areas by 2050 for major land-demanding interventions of the sustainability agenda and major BAU other land demands.

	Current ~2020	Projected			Total			Perc. of Earth (2050)	Overlaps (2050)	
	(Mha) ^a	Mid	Low	High	Mid	Low	High		(Mha)	(%)
Spatialised interventions										
Wind onshore (space)	18.5	118	15.7	235	137	34	253	1.05%	44.3	32%
Solar off-roof	0.47	11	7	36	11.4	7.5	36	0.09%	4.8	37%
CSP	0.02	0.33	0.21	0.63	0.36	0.23	0.65	0.00%	0.14	36%
ETM (mining)	4.96	131	131	131	136	136	136	1.04%	42.2	31%
Bioenergy (BE) conventional	~75	-65	100	800	10	100	800	0.08%	3.1	30%
BE lignocellulosic		130			130			0.99%	50.4	37%
Carbon-oriented restoration ^b	-	464	464	464	464	464	464	3.54%	11.8	3.9% ^c
Conservation ^d	2,360	1,828	1,828	1,828	4,188	4,188	4,188	32%	109.8	2.6%
SUBTOTAL	2,454	2,618	2,546	3,494	5,077	4,930	5,878	38.8%	133	2.7%
Non-spatialised interventions										
BE in forests	?	270	270	270	270	270	270	2.06%		
TOTAL	2,454	2,888	2,816	3,764	5,347	5,200	6,148	40.8%		
Other BAU land demands										
Built-up	162	80	28	131	242	190	293	1.8%	32	20%
Cropland	1,566	158	46	270	1,724	1,612	1,836	13%	471	30%
Grazing (+intensive)	1,480	158	0	315	4,842	4,681	4996	37%	459	31%
Grazing (-intensive)	3,201								1085	34%
Forestry ^e	1,982	112	51	172	2,094	2,033	2,154	16%	1154	58%
SUBTOTAL	8,391	507	125	888	8,898	8,516	9,279	68%	3,201	
Minimal land use	2,602								1,043	40%
Snow & Desert	2,104								692	33%
TOTAL	13,097								4,936	
TOTAL LAND DEMAND		3,394	2,941	4,652	14,244	13,716	15,427	108.8%		

a/ Current areas from Openstreetmap for Wind (Reference year: 2024), Kruitwagen et al for Solar (2018), Guru et al for CSP (2023), Maus et al and Owen et al for ETM (2019), UFOP for bioenergy crops (2022) and WDPA including protected areas and OECM (2025) (cf. text S1.3); other current land uses from (Matej et al., 2025) (cf. text S1.4).

b/ Medium estimates are the average of high and low for BAU land demands, and the retained target for the interventions. For some interventions, only the mid value was used for low and high estimates (in grey)

c/ Includes afforestation. This includes 160 Mha pledged by Saudi-Arabia in other countries which were not spatialised.

d/ Percentage calculated with the spatialised restoration area (i.e. 304 Mha of the total 464 Mha).

e/ Includes ~160 Mha of currently protected land in countries already having >30% of their land protected.

f/ Only includes timber.

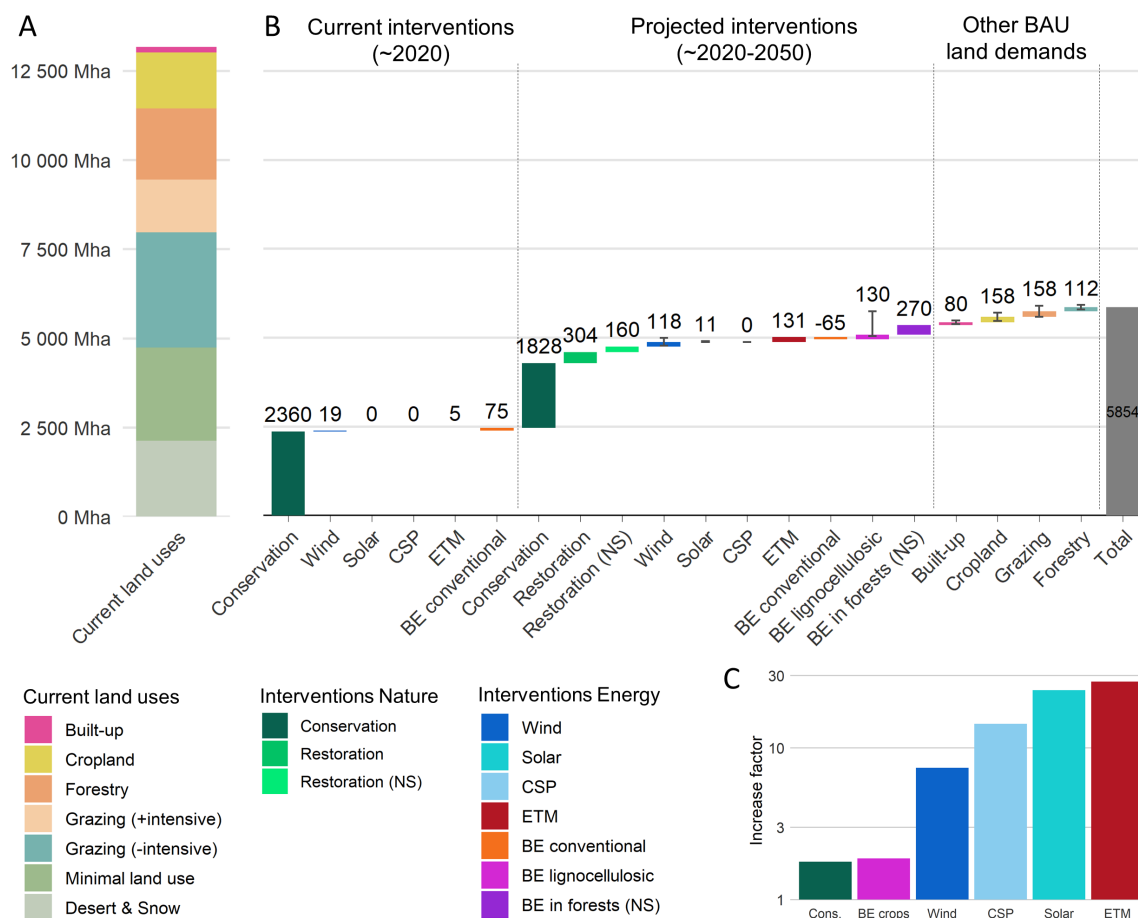


Figure 4.1. Required land area and expansion for the considered interventions and major land uses by 2050 in Mha. (A) Current (2020) land uses from (Matej et al., 2025), put aside (B) the current area occupied by the considered interventions (around 2020), their projected additional land requirements, and the other major BAU additional land demands foreseen by 2050 from existing sources, with ranges between low and high estimates (cf. table 1). (C) The expansion rate of each intervention for which we have a current area estimate (see Figure S1 for the absolute area expansion). NS: not spatialised.

The rate of expansion of these interventions is extremely large, suggesting tensions and drastic transformations in landscapes concentrating them. The 1828 Mha increase of area protected (+77%) required to reach the 30% target corresponds to almost doubling an already important area. For carbon-oriented restoration, an estimated 464 Mha were pledged where a change in land use is expected, consisting mainly of large-scale reforestation and afforestation projects (399 Mha explicitly pledged, and 65 Mha estimated for countries having no explicit split of their pledge yet in 2023).

Non-biomass renewable energy technologies and ETM mining have massive expansion rates, with a required increase corresponding to over 6 times the initial area for wind turbines (118 Mha), 20 times for solar PV (20 Mha), and 10 times for CSP (0.33 Mha). This expansion would in turn depend on a large expansion of

mining area worldwide, estimated at 131 Mha, over 27 times the initial area – based on a 10-km buffer around each ETM mining site geolocation likely to be in operation in 2050 (cf. Methods). This reflects the foreseen demand for critical minerals for these technologies, set to quadruple by 2050 (IEA, 2022), mainly for electricity networks, electric vehicles, and battery storage.

According to what is presented as a ‘land-sparing’ scenario by IEA, bioenergy would require 140 Mha for conventional and lignocellulosic (also called 2nd generation) crops, i.e. 9% of current global cropland, plus 270 Mha of forests, i.e. 6.7% of the 4060 Mha of global forests (FAO and UNEP, 2020).

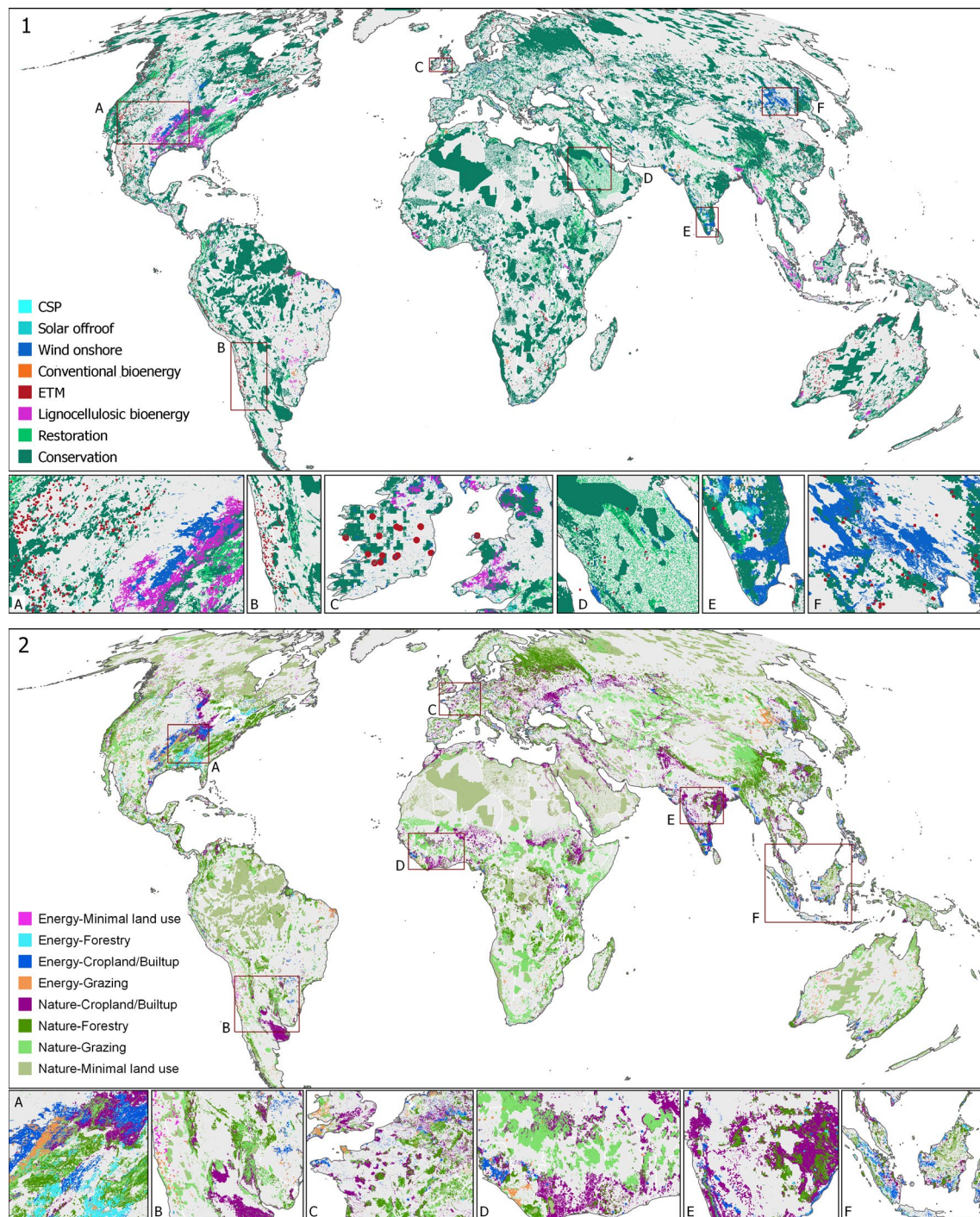


Figure 4.2. Maps of the 2050 spatialised interventions and their overlap with current land uses. (1) Distribution of each target/pledged intervention spatialised individually following the best-suited/priority land. (2) Overlaps between current land uses (only 1x1km pixels with >25% coverage of the considered land use) and 2050 spatialised interventions, by category with 'Energy' gathering Wind, Solar, CSP, ETM and bioenergy interventions, and 'Nature' gathering Restoration and Conservation. Maps were overlaid as shown in the legend (Conservation below, and CSP on top for A, and Nature-grassland below, and Energy-cropland/builtup on top for B), thus top layers can hide layers underneath.

4.2.2. Geographic patterns, and increasing pressure on existing land uses

We spatialised 4931 Mha, i.e. 92%, of the total 2050 required area for the considered sustainability interventions, including 2259 Mha already occupied by these interventions in ~2020 (17.25% of the ice-free land surface). We did not spatialize the current ~75 Mha of energy crops, nor the 270 Mha of bioenergy from forestry in 2050 and the 160 Mha of restoration pledged by Saudi Arabia in foreign countries (cf. Methods). The resulting geographic patterns show increasing pressure on land use and ecosystems worldwide, with hotspots in certain regions (Figure 4.2A).

Conservation is spread across all countries and biomes, per our approach. Hotspots of carbon-oriented restoration include the Eastern United States and North America's West Coast, the Middle East and Eastern Africa, and mountain regions of South and Southeast Asia. Some hotspots of lignocellulosic bioenergy include the Southeastern US and Insular Southeast Asia. Hotspots of onshore wind include Central-Eastern US, Northeastern China, and Southern India. Areas with important density of ETM mining include the Andes, Western US, Ireland, and Australia.

We assessed the overlaps between our projected interventions and current land uses, categorized as built-up land, cropland, grazing land, forestry land, and other land with minimal land use, i.e. areas with no or minimal discernable signs of these land uses (called 'wilderness' in (Matej et al., 2025)). 471 Mha (i.e., 30%) of current cropland would be impacted by the foreseen interventions in 2050 when spatialised using the best-suited land (Figure 4.3). 4/5th of these overlaps come from interventions to conserve or restore nature (78%), the rest divided between bioenergy (6.9%) and renewable energy technologies, including mining (8.4%). 31% of the 1480 Mha of more-intensive grazing land would be impacted, 33.9% of the less-intensive grazing land, 58.2% of forestry, 19.8% of the built-up area, and 40.1% of the area with minimal land use.

Energy-related land demands overlap with 4.6% of current cropland, 5.9% of grazing land, 15.7% of forestry, and 0.5% of areas with minimal land use. These interventions, targeting predominantly lands with adequate infrastructure and electricity grids, are in general more likely to come into friction with other human land uses than with remaining nature.

Some areas with high overlaps between interventions and specific land uses include Eastern and Central US (between energy and restoration, and agriculture and forestry); Argentinian pampa, West Africa and the Sahelian regions including

Ethiopia, and India (between nature interventions and cropland); throughout Sub-Saharan Africa (between nature interventions and grazing); parts of Europe and Insular Southeast Asia (between energy and cropland); and Southeast and East Asia (between nature interventions and forestry) (Figure 4.2B).

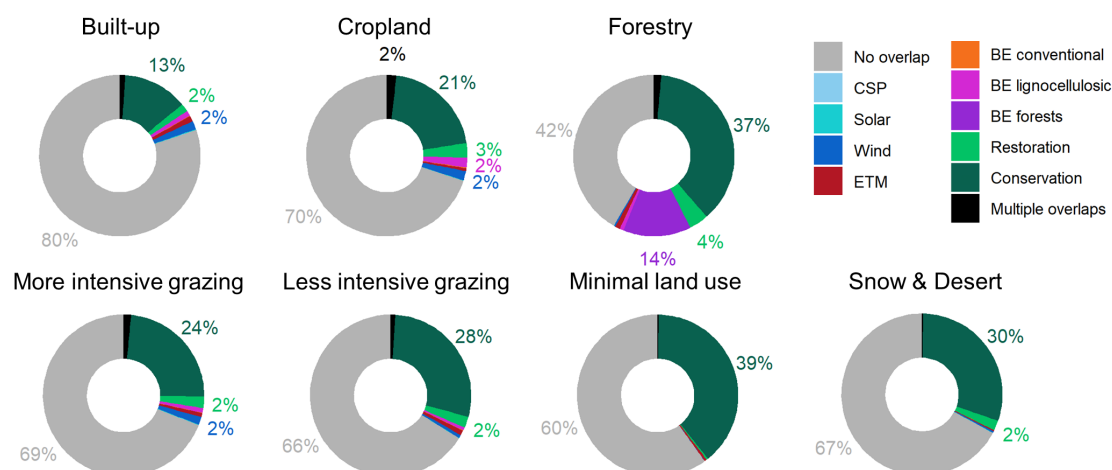


Figure 4.3. Share of current land use areas overlapping with 2050 interventions. Grey: no overlaps; color: overlap with one intervention; black: overlap with several interventions. ‘BE forests’ corresponds to the bioenergy sourced from forests according to the IEA 2050 Net-zero scenario, which necessarily overlaps with forestry activities. Percentages shown if 2%.

4.2.3. Overlaps between interventions

Overall, the total area of overlap between interventions reached 127 Mha, i.e., 2.7% of the combined area occupied by these interventions (4794 Mha), with large heterogeneity between interventions (Figure 4.4). This overall figure is pulled down by the large area dedicated to conservation and restoration, which cannot overlap as we only consider restoration involving land use and cover change. Yet, while these interventions have respectively only 2.6% and 3.9% of their 2050 area at risk, it still represents 110 Mha and 12 Mha, mostly overlapping with lignocellulosic bioenergy. The share of conservation is massive (85.5% of the total area of interventions, and 74.1% of the total intervention expansion), and it thus unsurprisingly drives 82.4% of the overlaps of the other interventions, compared to 17.6% induced by other interventions.

Other interventions have thus larger overlaps: energy interventions have on average 33.7% of their 2050 area overlapping with at least one other intervention, with individual interventions ranging between 30-37%. Expectedly, these overlaps occur mostly with conservation (~22-27%), but wind energy and lignocellulosic bioenergy also bring notable overlaps with the other interventions.

Some hotspots of overlaps between these interventions include Southeastern US and Insular Southeast Asia (between conservation/restoration and bioenergy), Europe, India and Northern China (between conservation/restoration and electricity production), or the Andes (between conservation/restoration and mining) (Figure S2).

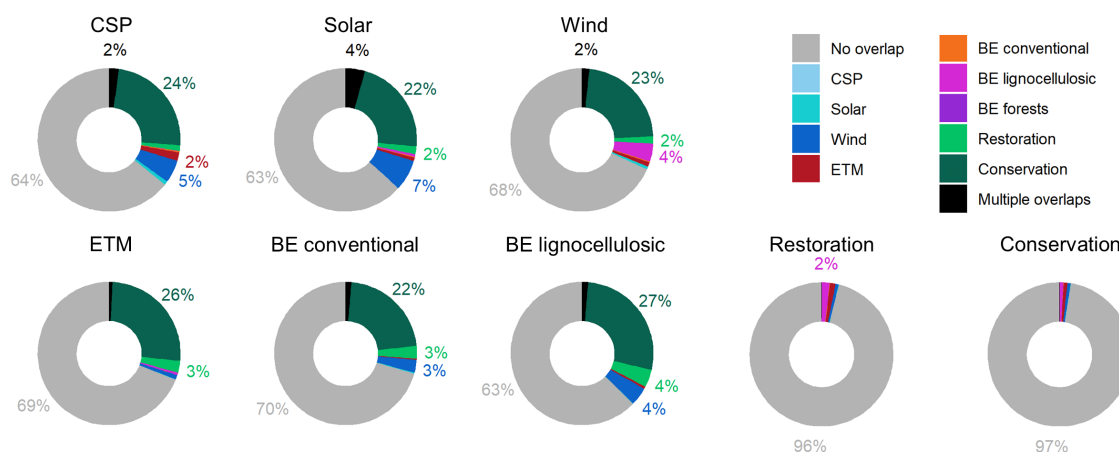


Figure 4.4. Percentage of the 2050 interventions overlapped by other interventions in 2050. Grey: no overlaps; color: a single overlap; black: overlaps between more than two interventions. Percentages shown if 2%.

4.2.4. Unequal land wealth

These global results translate into very different country realities. Of the 222 countries and dependent territories larger than 100 km², 72, encompassing 56% of the global human population, experience a 2050 land demand for climate and biodiversity interventions that reaches over 35% of their land area (including the 30% conservation required as part of the CBD pledge, and over 5% of their land area required for other interventions). Fourteen countries, representing 2% of the population, even have over 50% of their land pledged for climate and biodiversity, noting that this does not include land use for other basic needs like food and shelter.

The top-down suitability-driven siting applied here, assuming no cross-sectoral coordination, leads to hotspots of overlaps with existing land uses in certain regions (Figure 4.2B). Some countries have high percentages of their 2020 cropland, grazing land or forestry overlapping with at least one intervention in 2050 (Figure S3). This includes 127 countries (representing 77% of the global population) who have >25% of their cropland area under overlaps, and 20 countries (5% of population), including Ethiopia, Egypt and Venezuela, with >50% of cropland under overlaps. 178 countries with grazing land occupying over 10% of their surface

(93% of the population) have >25% of their grazing area under overlaps, and 20 countries (7%) >50%, including countries such as Bangladesh, Tanzania, Thailand, Turkmenistan, and the UK. Similarly, 129 countries with over 10% of their land used for forestry (71% of the population) have >25% of their forestry area under overlaps, and >50% for 42 countries (51% of population) including India, China, Germany and Venezuela.

Finally, 21 countries (home to 8% of Earth's land and 24% of the population) have >5% of their area with minimal land use under overlaps, not accounting for conservation, including for example Uganda, India, Kazakhstan and Denmark.

Some countries have high shares of overlaps between interventions themselves, due to either limited suitable land, or spatially coinciding suitability (cf. next section). In 36 countries, >5% of the area identified for interventions has overlaps between these interventions (noting again that this is dragged down by the large and non-overlapping conservation and restoration areas). 40 countries have >50% of their wind-targeted area overlapping with at least another intervention, and 38 countries for solar PV with the same threshold, 38 for CSP, 43 for bioenergy, and only 1 for restoration and 2 for conservation.

4.2.5. Suitability trade-offs

For identifying priority areas in each country, we ranked pixels in terms of suitability for each intervention (see Methods). Most countries hold sufficient land with reasonably good 'suitability' (or 'priority') to absorb the required areas, i.e. the average suitability of the lowest percentile selected over all considered countries ranges between 0.61 (for conservation) and 0.92 (for BE conventional) (Figure S4). Yet, some countries have to commit land with lower suitability. For solar as an example, in 20 countries, the average suitability of the lowest percentile selected is below 0.6 (5 countries for wind, 15 for CSP, 1 for BE lignocellulosic, 30 for restoration and 70 for conservation). Several countries even lack sufficient suitable land for specific interventions: 25 countries for wind, 6 for solar, 20 and 12 rather small countries (mainly island states) for restoration and conservation.

An alternative to spatializing interventions separately would be to distribute the interventions by avoiding the regions with overlaps with other interventions. To assess this, we masked out the areas targeted by any intervention, and calculated, for each intervention, the average suitability of the remaining areas, excluding those with no potential for that intervention (thus excluding completely unsuitable areas). This would imply to target regions with markedly lower suitability, ranging

between 0.46 (for conservation) and 0.67 (for wind). Across interventions, the average suitability of areas without any spatialised intervention is 24% lower than the suitability of the lowest percentile selected in the main scenario (Figure S4); for example, the average suitability for wind drops from 0.80 to 0.67. Such drops in suitability would mean generating lower yields for energy production and less biodiversity richness and carbon captured for conservation and restoration – thus requiring more land to reach the overall return targeted.

4.3. Discussion

4.3.1. Growing land demands on a finite planet

Overall, when compiling all commitments for 2050, 30% of Earth's land would be dedicated to conservation, and ~9% for renewable electricity, bioenergy, mining for ETMs, and tree planting for carbon storage, involving massive land use changes. On top of these, growing populations and per-capita income would entail a ~510 Mha (3.9% of Earth's land) expansion of agriculture, built-up land and forestry for timber by 2050 under BAU scenarios, with a large potential range mostly coming from land dedicated to animal products. Yet, estimates suggest that much less than 450 Mha might be available for cropland expansion in unforested, unprotected, low population density areas with low social and environmental tradeoffs (Eitelberg et al., 2015; Lambin et al., 2013). Together, all land demands for 2050 amount to ~14,000 Mha, to fit into a ~13,100 Mha ice-free land area. This suggests that achieving all these targets, including baseline food and fiber demand, would be impossible without massive efforts to organize synergies among overlapping land uses across thousands of millions of hectares.

We did not include some other land demands, such as non-ETM mining or other non-BAU projections like a surge in wood demand for construction if cement and steel were to be substituted in 'bioeconomies' (Searchinger et al., 2023), and other pressures. About a quarter of the global land is already affected by human-induced degradation through climate change and unsustainable land management (which we do not account for due to high spatial uncertainties), which will increasingly hamper the capacity of many ecosystems to meet people's demands (IPCC, 2022b). Rising sea-levels threaten ~105 Mha of highly populated and fertile land (Hooijer & Vernimmen, 2021), increasing inland land-demand for displaced populations and land uses (Hauer, 2017). Urban sprawl occurs on land more fertile than the average, so the displaced cropland requires larger areas of less-fertile land to compensate for the productivity loss (Bren d'Amour et al., 2017).

About a third of current cropland and grazing land, and over half of the land used for forestry, would be targeted by the considered interventions if implemented following a top-down and siloed approach. Over a third of energy-related best locations are also targeted by other interventions, in particular the conservation target to meet CBD commitments. Large disparities exist among countries and regions, with some countries having >50% of their current land uses potentially targeted by sustainability interventions. Some areas are expected to be hotspots of overlaps, with high competition. This reflects that all land does not hold the same value for expanding land uses, with some areas attracting high competition, e.g. solar PV units target mainly flat, accessible, and sun-bathed land, thus directly competing with agriculture (Adeh et al., 2019; Prados, 2010). To avoid overlaps, some interventions would need to be sited on less favorable locations, thus increasing the land required or reducing outcomes.

4.3.2. What do these overlaps represent?

Overlaps between interventions and with other land uses bundle complex realities. Different combinations of land uses in the same place can vary from antagonistic to synergistic, depending on the assessed outcomes and the context. When focusing on the main intended outputs (e.g.; agricultural production for cropland, biodiversity for conservation areas), some overlaps might be considered as mere coexistence, e.g. wind mills might be interspersed within cropland with minimal impacts on yields (Kaffine, 2019). Certain overlaps are almost necessarily exclusive, e.g. intensive bioenergy crops with conservation or restoration (Lewis et al., 2019; Searchinger et al., 2023). Other overlaps would involve trade-offs, e.g. wind farms might coexist with restoration or conservation areas, though likely reducing their biodiversity value (Gasparatos et al., 2017). Some combinations, such as agrivoltaism, which co-locates agriculture and solar energy production, requires tailored approaches, having so far mixed outcomes (Ali Abaker Omer et al., 2025). Still others might involve synergies, e.g. co-siting solar and wind infrastructure can harness both winter winds and summer sun, maximizing annual productivity (Qiu et al., 2024), and conservation measures in farmland can support both biodiversity and agricultural production (Cozim-Melges et al., 2024; Ricciardi et al., 2021).

When adopting a broader perspective on social and environmental trade-offs and equity (Mehrabi et al., 2018; Schleicher et al., 2019), these overlaps and their potential indirect rippling effects become even more complex to apprehend. Overlaps already exist today, providing clues on dynamics that may exacerbate; e.g. we estimated that ~690 Mha (~11%) of current cropland or grazing areas are located in

wind or solar farms, in conservation areas, or in the surroundings of mining projects. Solar or wind projects on farms can represent additional income for land-owners but also impact on farmers' access to land via increased land prices (Vez-zoni, 2025), and may involve trade-offs with other values such as cultural landscapes (Gibbons, 2015). Large-scale mining investments may displace populations and impact food security, soil and water pollution (Giljum et al., 2022). Conservation interventions on populated lands have to confront entrenched legacies of dispossession, marginalisation and prejudice (Adams & Hutton, 2007). Overall, social-ecological contexts and governance and equity issues are often inadequately addressed when implementing interventions for conservation (Lele et al., 2010), restoration (Malkamäki et al., 2018), or low-carbon transition (Sovacool, 2021), thus triggering challenges and tensions, including land conflicts and dispossession, environmental degradation, and social unrest.

These dynamics are expected to exacerbate as an additional 1,320 Mha of the ~6250 Mha of the 2020 crop or grazing land would be affected by one or several interventions. This is already evidenced by the rise of 'green grabs'; the appropriation of land by governments and large companies to implement interventions linked to the global sustainability agenda, excluding local and small-scale land users to the benefit of more powerful actors (IPES-Food, 2024). Moreover, many climate and biodiversity initiatives are promoted by the Global North, but to a large extent outsourced to the Global South, such as carbon offsetting (Dooley et al., 2024; Fairhead et al., 2012). This risks contributing to further entrenchment of systemic unequal ecological exchange (Givens et al., 2019).

4.3.3. The need for breaking silos and centring equity-concerns

Policy-makers tend to silo individual issues separately (Pörtner et al., 2021), with compartmentalized institutions (Bazerman, 2006). Yet, the climate and ecological breakdown and the growing global land scarcity are tightly entangled with multiple other 'wicked problems' such as rising inequality (Chancel et al., 2022), prevalent undernourishment (FAO et al., 2024), resource depletion and pollution (UNEP, 2024b), democratic backsliding (Lührmann & Lindberg, 2019; Meyfroidt, 2025), or power concentration in the hands of a small number of multinational corporations (Clapp, 2021). Single-focus policies can improve on specific indicators while negatively impacting others (Prudhomme et al., 2020). Understanding and addressing these entangled crises requires comprehensive approaches (Lawrence et al., 2024), including negotiating targets across different domains, analysing involved trade-offs locally and globally, and strengthening multi-stakeholder coordination

in the spatial planning and implementation of sustainability interventions (Díaz et al., 2020). Regulatory schemes are critical to prevent land use expansion to merely follow market forces, targeting lands with the highest potential for production without considering tradeoffs (Brunelle & Makowski, 2020). Claims that new land uses such as energy crops or tree plantations are deployed on ‘marginal lands’ to reduce trade-offs have proven elusive so far (Crawford et al., 2022). For research, this involves holistically considering socio-economic and governance conditions to accurately represent local realities (Benra et al., 2024). Overall, wearing an equity lens when assessing and implementing these ‘green’ interventions is critical, centering on the participation, engagement and well-being of local and most-affected people and communities (Löfqvist et al., 2023; Rot et al., 2025). We advocate a shift in perspective, from analyzing individual trade-offs and impacts to integrating societal and ecological objectives on an equal footing (GSDR, 2019).

4.3.4. How much can these targets be adjusted?

Taken together, the area needed to host these interventions as well as current and BAU land use demands is larger than Earth’s ice-free land surface. We thereby consider here whether our assessment under- or overestimates these land demands, or whether they could be reduced.

The land requirements for conservation are large, though some advocate for even more (Dinerstein et al., 2020; Wilson, 2016). Smaller overlaps and lower competition for land might be achieved by concentrating conservation areas in regions with low population and productivity, e.g. remote tropical forests or boreal regions. Yet, spreading conservation across biomes and countries is necessary to preserve representative biodiversity and the delivery of local nature’s benefits to people. That said, our spatialization leaves vast areas unprotected in highly biodiverse and carbon-rich areas such as the Amazon and Congo basins, although other international commitments (Glasgow Declaration, Global Stocktake) prohibit forest conversion, notably to stay within the Paris Agreement. The 30% global objective will presumably balance between representative conservation, as we simulated, and focused efforts towards critical regions. Mitigating trade-offs will require to adequately mobilize a diversity of types of protected areas, with gradients of inclusion of human activities, noting that under certain conditions, protected areas have shown positive effects on local people’s livelihoods (Oldekop et al., 2016).

Overall, evidence shows that land-based interventions such as tree planting and bioenergy expansion specifically for carbon dioxide removals (CDR) require large swathes of land for little effectiveness, and can lead to counterproductive effects

(McLaren, 2020). Yet, many countries pledged to increasingly rely on CDR to achieve net-zero emissions (Hale et al., 2022; Schenuit et al., 2021; Vaughan et al., 2024), including through enhanced land carbon sinks (Dooley et al., 2024; Smith et al., 2022). Carbon-focused restoration with land use conversion comes second in our assessment of land demands, with an estimated area of 460 Mha, much higher than the 195 Mha reforestation potential identified when accounting for previous critiques on the social and environmental impacts of reforestation projects (Fesenmyer et al., 2025). Private companies' land-based pledges to compensate for their emissions remain scarcely documented (GRAIN, 2024), but are likely large – with ~60 Mha pledged by only four energy firms (Sen & Dabi, 2021). Despite being in principle recorded in countries' accountings, they might further increase pressure on land. 'Restoration' can take many forms; from natural forest regeneration to plantations of commercial trees. Reorienting pledges from plantations of fast-growing trees (Dooley et al., 2024; Lewis et al., 2019) towards restoration of natural ecosystems would, over the long term, store more carbon, provide a broader range of nature's contributions to people (Lewis et al., 2019), and achieve synergies with conservation (Chapman et al., 2025).

BECCS is also promoted as a CDR option, such as in IPCC scenarios (IPCC, 2022a; Vaughan et al., 2018), with estimates of energy crops' land requirements in 2100 ranging from 200-800 Mha for 2°C and 1.5°C pathways (Creutzig et al., 2021), i.e. 13 to 52% of current global cropland. Bioenergy (with and without CCS) comes third in our land demand assessment (410 Mha in 2050), although we used a scenario presented as low-land demanding, with 'only' 9% of current cropland required. Research has increasingly demonstrated the land-use trade-offs of bioenergy, particularly with food production and natural ecosystems (Gerssen-Gondelach et al., 2017; Humpenöder et al., 2018). Bioenergy expansion induces spillovers that increase deforestation and emissions in distant locations (Guye, 2025), leading to possibly higher emissions than fossil fuels (Merfort et al., 2023). Bioenergy crops require around 30 times more land than solar PV to produce an equivalent amount of energy (Sturchio et al., 2025). With expected exacerbated pressures and competition for land, bioenergy does not appear as a desirable pathway to mitigate climate change. Further, relying on land-based CDR risks delaying fossil fuel mitigation efforts (Allen et al., 2022; Doelman & Stehfest, 2022), as manifest in country pledges being inadequate to meet the Paris Agreement (UNEP, 2024a).

The deployment of solar and wind energy infrastructures is key to shift away from fossil fuels, but the large land tracts required, including for mineral extraction, brings strong implications on nature and people. With 285 Mha occupied or

affected, renewables come fourth. This includes the area directly covered as well as under influence of these infrastructures; i.e. including space between wind turbines, and land within a 10 km buffer around mining geolocations, which might be affected in multiple ways (deforestation, pollution, erosion) (Giljum et al., 2025; Sonter et al., 2020). As a comparison, IRENA (2019) forecasts 100-166 Mha of onshore wind footprint for 5044 GW in 2050 (vs. 118 Mha in our scenario, for 7114 GW). In such a 100% WWS world, some land would be released through fossil fuel and uranium phase-out, but our assessment did not include electricity networks, battery storage (King et al., 2023), and industrial requirements (for construction, recycling, and waste landfills), while uncertainties remain on many aspects including the life span and recycling rate of minerals used (Watari et al., 2020). Extractive activities leave environmental and social impacts well beyond their closure (Sovacool et al., 2020). Further, so far, the steep rise in solar and wind energy deployment has not led to an absolute reduction in fossil fuel use; energy use remains very high in the Global North and rising globally (EI, 2025). At this rate, under a BAU scenario, the combined footprint of coal, conventional oil and gas would increase by 97 Mha by 2050 (Johnson et al., 2021). To avoid putting pressure on land without effects on climate change mitigation, future deployment of renewables has to lead to actual reduction in fossil fuel use.

4.3.5. Efficiency

Another proposition to mitigate land pressures is increased efficiency, or intensification, i.e. delivering more outputs from the same land area. Intensification has a role to play in reducing pressure on land (i.e., “land sparing”) (Balmford et al., 2025), but appears insufficient, as its effectiveness is partly curtailed by rebound effects (Rodríguez García et al., 2020). The success of intensification in reducing natural habitat conversion is highly context- and crop-dependent, but, overall, current rates of productivity growth are insufficient to avoid cropland net expansion at global scale (Villoria, 2019).

Further, the net environmental impacts of intensification depend on the boundaries – spatial, temporal and sectoral – used in these calculations, often revealing secondary effects (Styles et al., 2018); e.g., intensive land management can increase fertilizer and pesticide run-off (Clark et al., 2018), negatively impact biodiversity (Semenchuk et al., 2022), and contribute to climate change (Xing et al., 2024), and technology efficiency often rely on limited, socially and environmentally impacting ETMs (Depraeter & Goutte, 2023). More intensive systems are also increasingly sensitive to perturbations, e.g., double-cropping in Brazil (Xu et al., 2021) is

now threatened by climate change (Rattis et al., 2021). Compensating for land degradation and crop failures could, in turn, require further land expansion.

Sustainable intensification is increasingly proposed as a middle-of-the-road option, yet, improving production management cannot address all trade-offs. Overall, growing evidence shows that technological innovation (efficiency) alone will not enable economies to stay within planetary boundaries, pointing to the need for also managing demand and consumption (Vadén et al., 2020).

4.3.6. Pressure on land cannot be mitigated without reducing land demands

Our findings point to a few conclusions: (i) Without overlaps, there is just physically not enough space to accommodate for all land demands, (ii) Some overlaps can be neutral or beneficial, but in many cases they involve trade-offs, (iii) Some of these land demands appear hardly compressible without harming nature and societies, like conservation, or a sufficient amount of non-biomass renewables to displace fossil fuels, (iv) Increasing efficiency, if well-managed, plays an important role, but also brings its own trade-offs and risks.

Together, this leads to the implication that sufficiency scenarios for high-income populations are crucial to reduce land requirements and environmental pressure, and mitigate trade-offs, while allowing to meet basic human needs ensuring an equitable well-being (Carmenta et al., 2023; Erb et al., 2024; O'Neill et al., 2018; Raworth, 2017). This entails both demand- and production-side changes focusing on affluent countries, companies and populations, as wealth plays a predominant role in biodiversity decline and climate change (Otero et al., 2020; Schöngart et al., 2025; Wiedmann et al., 2020).

Key levers to reduce land pressure include dietary changes, such as meat consumption reduction (Leclère et al., 2020; Theurl et al., 2020), and reduced production and consumption of material goods which would ultimately decrease energy requirements, with cascading effects on lessening demand for land-based carbon sinks, bioenergy, electricity production, and mining (Millward-Hopkins, 2022). Several alternative perspectives have proposed a profound rethinking of economic development models, such as doughnut economies (Raworth, 2017), democratic economic planning (e.g. Devine 1988) or degrowth (Weiss & Cattaneo, 2017), opening ways to systemically address overconsumption and production. Yet, these alternatives still remain to be translated into sufficiency targets and policies within the global sustainability agenda.

4.3.7. Limitations

First, uncertainties arise from the quantification of targets and their conversion into areas (cf. methods). The renewable energy dataset (Jacobson et al., 2022) covers 149 countries (although 97% of the world population); other countries' energy spatial footprint is not assessed, but expected to be very small. We used fixed density factors to convert installed capacity, and fixed buffers around mining sites, while real-world situations vary widely. We included all mining sites extracting or projecting to extract ETMs, while some might in fact never start operating — especially in densely populated areas. Finally, land requirement estimates for carbon capture or fossil fuel substitution through restoration, afforestation and bioenergy highly depend on the approaches used, and policy demands and consumption scenarios (Climate Resource, 2023; IEA, 2021). For countries without public pledges, we applied the average share of pledges involving land use change from other countries, hence, our total restoration/afforestation number is not an actual government figure. Overall, improving transparency and consistency in country and company pledges is urgently needed to allow for robust assessments of their implications on land and people (Dooley et al., 2024). We did not account for hypothetical technological breakthroughs which could reduce the land demand of these targets.

Second, we rely on suitability and priority assessments which come with built-in limitations. The suitability dataset (Oakleaf et al., 2019) we used for the spatialisation of wind turbines shows very little suitability across several regions, which mainly comes from lack of grid infrastructures, and thus might change in the future. Importantly, the suitability maps represent the current situation, but actual suitability might have changed in 2050, e.g. infrastructures for the electricity grid are likely to differ, and climate change will likely have decisive impacts influencing the biophysical potential and thus siting of the considered interventions.

Third, we made specific choices which influence our results. We split conservation targets by country and by biome. We did not map offshored restoration pledges (e.g., Saudi Arabia's pledge for Middle East countries), nor forestry-based bioenergy. Allocating land targets using best-suited pixels also comes with limitations (cf. text S1.1). We did not account for equilibrium effects – i.e. dynamic land competition which would reallocate land use. Future work could explore different ways to distribute these pledges, e.g., achieving 30% of conservation globally rather than by country and biome (Jung et al., 2021).

4.4. Conclusion

We compiled and spatialised targets from the global sustainability agenda and having implications on land. We show that achieving all these targets, while meeting baseline food and fiber demands, would require tremendous efforts to mediate trade-offs among overlapping land uses. We stress the need to move beyond siloes towards a systemic approach to sustainability which navigates trade-offs by centering equity concerns, and ultimately seeks to minimize these trade-offs by reducing overconsumption and production.

Top-down mapping yields considerably different results than bottom-up land-use planning (Venier-Cambron et al., 2023; Windey, 2020, p. 202; Wolff et al., 2020). A fairer spatialisation of these pledges would be ‘constructed from the ‘ground up’, drawing on a rich understanding of empirical context’ (Wyborn & Evans, 2021). Hence, we highlight again that this assessment is not meant as a forecast of what will happen, but rather as a thought experiment to raise awareness and foster dialogue on the limits of land.

4.5. Methods

We followed four main steps (Figure 4.5, Figure S5), described in detail in SI (text S1). First, we compiled quantitative targets, expressed through multilateral agreements or public commitments for each sustainability intervention, with target years either 2030 or 2050. Specific choices, presented in the introduction, are further detailed in SI (text S1.3).

Second, we compiled global spatially-explicit suitability and priority (hereafter: ‘suitability’) datasets (Figure 4.5), namely for conservation (Jung et al., 2021); restoration (Strassburg et al., 2020); renewable energy technologies (Oakleaf et al., 2019); ETM mining (Owen et al., 2022); lignocellulosic and conventional bioenergy (Li et al., 2020; Oakleaf et al., 2019). When necessary, we projected the input data to the equal-area Mollweide projection and resampled them at 1x1 km, by applying bilinear resampling method for continuous raster data and nearest-neighbor method for discrete data. The ‘current’ (i.e. ~2020) distribution of the considered intervention was masked to only consider areas for expansion. The following datasets were used for current distributions: for conservation (UNEP-WCMC and IUCN, 2025); wind turbines (from OpenStreetMap); solar panels (Kruitwagen et al., 2021); CSP (Thonig et al., 2023); mining (Maus et al., 2022). To assess overlaps with current major land uses, we used a recent global land use dataset (Matej

et al., 2025) for 2020 representing cropland, grazing land, forestry, built-up land and wilderness (which we renamed area with ‘minimal land use’) (text S1.4).

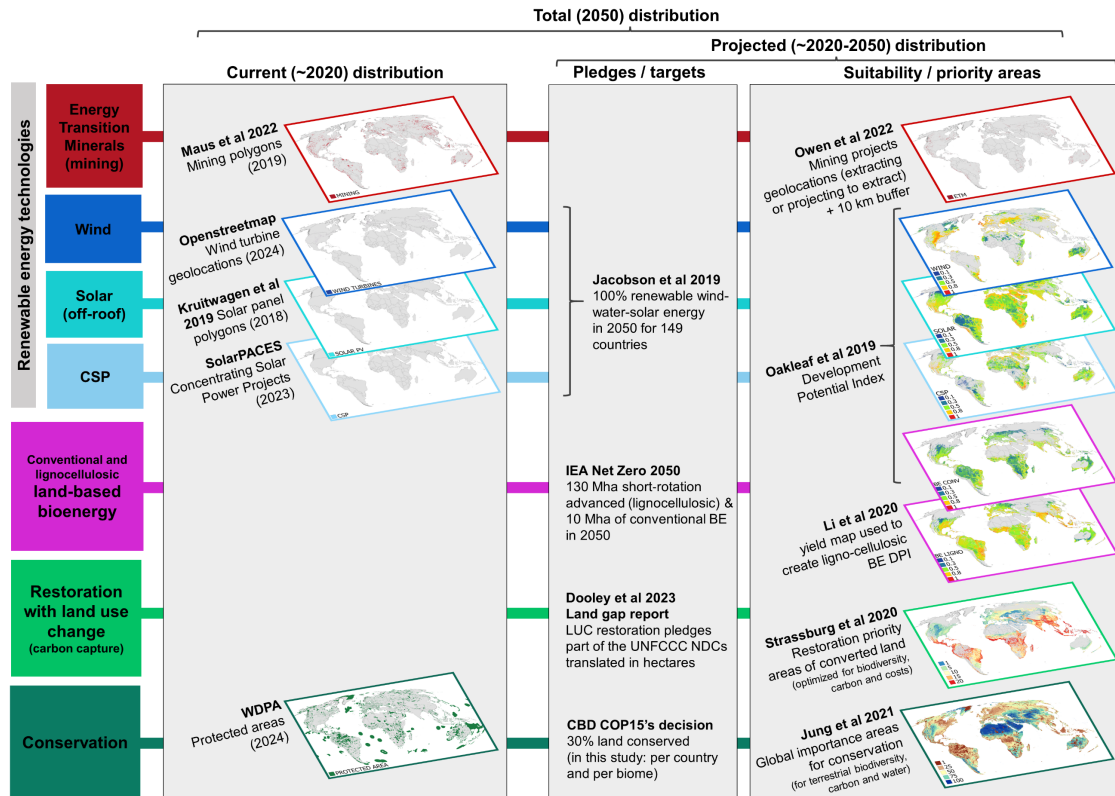


Figure 4.5. Datasets used and workflow. From right to left, starting with the collection of pledges or targets for each intervention, projecting them spatially using global suitability or priority maps to obtain their 2020-2050 area distribution, to which we combine the current distribution to generate the 2050 area distribution per intervention.

We normalised the suitability rasters by rescaling them from 0 to 1 per country. We reclassified the country-standardized suitability rasters in percentiles, per country. Then, we calculated the area per percentiles for each country for each intervention and selected the ‘best-suited’ percentiles to cover the target area (see text S1.1 and 1.2). We stress that the notion of ‘best-suited’ is limited to what these global maps consider, and does not capture elements that only local assessments could comprehend. We also calculated the mean of the non-normalised suitability values of (i) all the selected pixels and (ii) the last selected percentile, per country, to get an overview of the mean, and the lowest suitability/priority score required to fulfill the pledges – providing an idea of the ‘returns’ of the targeted areas.

Third, we overlaid the selected areas for each target and identified the share of each intervention’s targeted locations that overlap with that of another intervention, and with existing land uses and covers.

Fourth, we gathered recent literature estimates of other business-as-usual land demands (table S1), retaining high and low estimates by excluding outliers and poorly supported numbers, and scenarios or projections that include disruptive elements such as drastic policy, consumption or technological changes), and calculating the medium estimate as the average of the high and low estimates.

4.6. References

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Chapter 5

Discussion and perspectives

Several decades of sustainability talks, commitments, and action, have not prevented the climate and ecological breakdown from worsening. Here, after taking stock of the main findings and limitations of previous chapters (section 5.1), I reflect on the reasons behind the long-lasting failures of sustainability initiatives and the effects of ‘narrow and shallow’ approaches (section 5.2 ‘Blinders & silver-bullets’), then I delve into the root causes of the biodiversity and climate breakdown (section 5.3 ‘Searching for roots of intertwined problems’) and, finally, I try to look ahead and answer this broader question regarding sustainability issues: ‘What can we do about it?’ (section 5.4 ‘Tackling root causes’).

5.1. Synthesis

5.1.1. Main findings

The first three chapters explored the land demand outcomes and the limitations or risks of current sustainability interventions.

After having demonstrated the role of cocoa as a major driver of deforestation in Côte d'Ivoire and Ghana, with about half of the undisturbed tropical moist forest lost between 2000 and 2019 converted into cocoa, chapter 2 and 3 explored how traceability and zero-deforestation commitments, meant to secure deforestation-free cocoa supply chains, risk facing leakages. **Chapter 2** highlighted the complexity and the challenge facing traceability in the cocoa sector by showing that the majority of cocoa exports of 2019 from Côte d'Ivoire were either untransparent (sourced via traders that do not disclose any information) or indirectly sourced, with the origin unknown. It stressed the inherent limitation of traceability objectives focused on direct sourcing only, with almost 60% of the deforestation exposure linked to untraced sourcing. **Chapter 3** further demonstrated the limitations of 'narrow' sustainability initiatives restricted to a single sector like ZDCs. It showed that though conversion to cocoa in gazetted forests is limited, cocoa expansion and land saturation off-reserve indirectly displaces food crops in forests. About a fourth to a third of the forest land converted to food crops corresponded to food crops being displaced by other land uses – with half of this displacement being due to cocoa expansion. Accounting for both its direct and indirect impacts, cocoa is an important pressure on these gazetted forests and as cocoa saturates the space outside of remaining forests, this pressure is expected to grow.

It appeared that as long as sustainability policies remain individual company choices (i.e. ZDCs), or are enforced only for a single export-market (i.e. EUDR), the whole sector is not covered and farmers can continue to produce cocoa linked to deforestation while selling it to other buyers or through informal local intermediaries (chapter 2). Beyond the lack of complete 'intra-sector' coverage, 'inter-sector' coverage is also missing; i.e. as all land uses are not covered by such policy, uncovered crops may expand or be displaced in forested areas instead of the target commodity (chapter 3), preventing deforestation from stopping.

Land being limited, it begged the question of how much cocoa can actually be produced sustainably, free of deforestation, considering the multiple and growing human needs? In these landscapes, the same question could be asked for food crops, palm oil, or any other land use, and, ultimately, for *anything* requiring land. This question called for broadening the perspective, which **chapter 4** did by assessing at the global level major land demands foreseen by 2050. Together, land-

demanding targets and pledges relating to global climate and biodiversity issues, and major land demands for food, fibre and timber amount to ~14,000 Mha, to fit into a ~13,100 Mha ice-free land area. Sustainability interventions' best suitable locations overlap with about a third of current cropland and grazing land, and over half of the land used for forestry. Interventions would also partly compete for the same land, with over a third of energy-related best locations also targeted by other interventions, in particular the conservation target to meet CBD commitments. These global figures hide large disparities with some areas expected to be hotspots of overlaps, with high competition. Various overlaps can entail a variety of trade-offs²⁵, depending on the context and on the spatial, temporal, social and sectoral boundaries used in these calculations. Although some may work, ultimately, these sustainability targets would further increase the global land squeeze, with the risk of spurring green grabs.

Chapter 4 also discussed the limitations of the potential of these interventions to address the biodiversity and climate breakdown, especially bioenergy and carbon-oriented plantations, as well as land intensification. It called for breaking silos, and for a systemic approach to sustainability – ultimately seeking sufficiency to minimize trade-offs by reducing overconsumption and production. It emphasised that key levers include reduced production and consumption of material goods for the wealthy (Millward-Hopkins, 2022) and dietary changes, particularly through a reduction in the consumption of animal products. With agricultural land occupying over half of the ice-free land surface, and about 80% of it being dedicated to livestock farming, dietary change policies in meat-intensive countries would improve health while significantly reducing agricultural land requirements (Leclère et al., 2020; Theurl et al., 2020).

5.1.2. Limitations

Chapters 2-4 have limitations that can be synthesized as well. First, a fundamental limitation pertains to the reliance on remote sensing datasets. In each chapter, sets of global and local spatial datasets were combined, with each of them bearing their own errors. When piecing together diverse products, errors propagate, and thus one should find ways to acknowledge and make these errors visible such as through confidence intervals. This work benefitted from the good quality ETH cocoa map,

²⁵ This explains why we did not categorize these overlaps along a gradient of antagonistic-synergistic combinations, as it depends on the perspective – when including social and environmental trade-offs and equity, it becomes complex to classify them, raising a series of questions; e.g., What would be these categories? Which criteria would be used? Who decides which overlaps are 'good' or 'bad'?

and used the JRC dataset on tropical moist forest (TMF) which offers a great longitudinal dataset at global scale. Yet, TMF shows limited quality around the early 2000s in Western Africa, where clouds complicated satellite detection, bearing the risk of detecting cocoa as forest – thus artificially inflating our deforestation results. Chapter 3 took this into account by limiting what was considered as forest in 2000 to a tree canopy higher than 8 meters (though the quality of the map used for this is also limited), but this was not done in chapter 2 (though the quality of Earth Observation data in Côte d'Ivoire was deemed higher than in Ghana). In addition, with pixels of 30x30 m, a single large tree could be considered as a forest pixel, and thus the removal of shade trees or rehabilitation of pre-existing cocoa farms possibly classified as deforestation. Chapter 3, unlike chapter 2, took this in account by limiting forest to patches of at least 1 ha.

Second, the deforestation measures include both legal and illegal activities – which was highlighted notably by quantifying deforestation both on- and off-gazetted areas. However, chapter 2 did not distinguish legal and illegal deforestation in the sourcing of companies. In a context of disturbed rainfalls due to forest loss and worsening climate change, in countries with little forest left, losing more forest – either legally or not – is problematic. Yet, it does raise a wider question about how numbers of hectares of forest presented as 'lost' (despite potentially being part of a cyclically shifting agriculture) almost automatically invoke an imaginary of damage imposed on nature as well as moral judgements intertwined with patriarchal and colonial prejudices. Conservationists and ecologists have been struggling with this for a long time. Similar issues come about with top-down and global mapping exercises which, simply, do not reflect local realities (see chapter 4). Acknowledging this malaise without trying to escape it is certainly key to cultivate humility.

Third, drivers of land use changes are complex and deforestation or forest degradation rarely have a single underlying cause. In this research, the term 'direct' is understood as a direct spatial association between deforestation and the detected land use, i.e., they both occur at the same location. The three chapters did not capture the temporal dynamic, whereby other land uses could have occurred in the period between the deforestation or degradation event and the detection of land uses in 2019, or sequences of multiple land use changes could happen on a given land between 2020 and 2050 (chapter 4). Results in chapters 2-3 are however expected to closely reflect cocoa's direct contribution to forest loss and degradation because cocoa has been a major expanding crop over the studied period, and because of its propensity to seek forest rent, making it a typical frontier crop.

Fourth, underlying drivers, especially the role of economic growth in promoting deforestation and land demands – including land demands to fix the environmental

impacts caused by economic activities themselves (chapter 4), were discussed but no causal analysis was conducted. As such, our land balance model in chapter 3 is an approximation of the role of cocoa expansion in the displacement of food crops in forests. Specifically, as we show, scale matters in such approaches, with different scales of analysis putting the focus on different sets of and balances between direct and underlying drivers. This analysis was not conducted at a finer, sub-regional scale due to limitations in data, though it could have highlighted other dynamics.

Fifth, limited public disclosures by companies on their supply chains, and unclear or inexistent public data on land implications of sustainability pledges by countries, also constituted clear limitations, let alone having substantially complicated our work. As only publicly disclosed data was used in chapter 2 to reconstruct the supply chain's traceability, our resulting level of traceability is subordinated to the level of transparency of companies. A simplistic binary categorisation of *transparent* vs. *untransparent* companies was also used (i.e. disclosing vs. not disclosing data), not capturing details in the level of transparency. Though considered transparent, disclosing companies may in fact not publicly share all their direct suppliers, hence impacting on our results. Overall, traceability efforts are undermined by the fragmentation of existing data systems and indicators, the lack of data disclosure by public and private actors, and limited data quality control (Dontenville et al., 2022). In a data intensive era, one could expect to have answers for crucial questions; Where do things come from? What are their social and environmental implications? What are companies and states intending to do, and how would that impact people and nature? The lack of transparency and traceability remains a major challenge for sustainability.

Sixth, climate change was not quantitatively incorporated into the analyses of chapter 4, though its effects is already exposing millions of people to acute food insecurity and reduced water security (IPCC, 2023c). With every increment of global warming, very wet and very dry weather and climate events and seasons will intensify, with repercussions on flooding or drought, and on the widespread deterioration of ecosystem structure and function (IPCC, 2023b). This will affect infrastructure, agriculture and forestry, with impacts on carbon capture by plantations, energy production from bioenergy crops or wind and solar infrastructure, or, more fundamentally, on food production and water access – to name just a few. Major shifts in spatial suitability for the considered interventions are thus to be expected, drastically impacting on their distribution by 2050. Coastlines – where wind turbines suitability tends to be high – are particularly at risk, facing sea level rise and intensification of flooding and cyclones. Yet, the location and frequency

of these extreme events and seasonal shifts depend on many factors (IPCC, 2023b), making it extremely complex to assess how each of these changes and their combined effects will affect each intervention – noting that the extent and magnitude of climate change impacts are larger than previously estimated, with the difficulty of taking in account complex positive feedback loops and possible abrupt responses or tipping points, such as strongly increased Antarctic ice-sheet melt, forest dieback or abrupt ocean circulation changes (IPCC, 2023b). Further, climatic and non-climatic risks will also increasingly interact, creating compound and cascading risks that are more complex and difficult to manage (IPCC, 2023c). Consequently, as stated in chapter 4, the spatialization of interventions is not meant as a forecast of what will happen, but rather as a thought experiment, aimed at fostering discussions and concerted actions accounting for trade-offs.

Chapter 4 also did not account for other sustainability targets, central to the transformation of food systems, such as GBF target 10 aiming at enhancing biodiversity and sustainability in agriculture, aquaculture, fisheries and forestry, or target 7 on the reduction of pollution levels, including pesticides. They were not included due to the uncertainty in how they would translate in terms of area (with some land management practices more extensive than others) and because the level of livestock production is the major determinant of land demand for agriculture. As stated in the introduction of chapter 4, unlike integrated assessment models (which work under a lot of assumptions), the objective was to highlight tension points and possible overlaps that might arise from assessments made separately across interventions rather than identifying scenarios that reconcile these different demands. Hence, chapter 4 is an attempt to outline the problem and raise awareness, rather than setting out practical solutions – beyond highlighting the need for alternative economic models that reduce throughput.

Overall, this work showed that current interventions lack effectiveness because they are limited to specific land uses or market segments and to direct drivers, failing to address sustainability issues as systemic and with complex and structural root causes. Yet, previous chapters did not analyse these structural roots in depth – which next sections attempt to do.

5.2. Blinders & silver-bullets

The climate and ecological breakdown and the global land squeeze are intrinsically complex problems rooted in multiple interdependent dynamics across multiple (if not all) sectors (Kaiser & Patterson, 2025). They are symptoms and causes of other crises, as discussed in chapter 4. Arguably, we are facing an emergent *global*

*polycrisis*²⁶, whereby multiple crises interact and have cumulative effects on several systems (Lawrence et al., 2024).

Yet, sometimes framed as silver-bullets, sustainability interventions are designed for a specific sector independently from others, and tackle singled-out problems (Pörtner et al., 2021), e.g., cocoa-traceability and increased productivity to curb deforestation, bioenergy and plantations to stop climate change – while these problems often are the symptoms of complex, unaddressed causes (Wise et al., 2014). This is exemplified by current multilateral environmental agreements which not only suffer from insufficient implementation, but also seldom establish clear links with the actual sources of the problems (Moranta et al., 2022). According to a recent extensive literature review conducted by IPBES, there appears to be ‘*a preference for so-called ‘technofixes’ or palliative measures that focus on minimizing direct drivers or mitigating their impacts while ignoring the underlying causes*’ (Gurung et al., 2025). Leach et al. (2010) have shown that there is a tendency to reconfigure uncertainty into calculations of risk and cost-benefit equations, ultimately ‘closing down’ to a reductionist frame that brings forth inappropriately narrow sets of measures. The resulting solution space is limited and largely unsuited to address root causes of problems (Wise et al., 2014).

Policies interventions designed for a given target sector, market segment, or spatial area could be described as ‘narrow’, and those that only focus on proximate causes of environmental harm as ‘shallow’ (Guye et al., *in prep.*). Narrow and shallow interventions can result from institutional and powerful actors protecting the status quo and their vested interests (Frantzeskaki et al., 2025; van der Ven et al., 2018), or from government departments and actors working in silos (Bazerman, 2006; Lah, 2025). Their prevalence may also be due to the fact that the world and its interrelated systems are simply too complex to apprehend fully. They may appear easier to implement with faster returns, while more systemic fundamental solutions may appear out of reach (Senge, 1990) and requiring more time, energy and hands-on commitment (Kim & Anderson, 1998). Ultimately, how problems are perceived and solutions imagined plays a critical role in the design of sustainability initiatives. Without often being aware of them, blinders limit our perception and imagination. These mental filters are constructed from beliefs, experiences, culture, education, and individual psychological development (Hayward, 2003) and highly depend on dominant paradigms (cf. section 5.3.2). Such filtering builds

²⁶ Defined by Lawrence et al. (2024) as ‘*the causal entanglement of crises in multiple global systems in ways that significantly degrade humanity’s prospects*’. This term captures how multiple interacting crises create harms far worse than the sum of the harms they would produce separately.

interpretations of how the world works, and can lead to cognitive biases, such as underestimating threats which are (relatively) slow to emerge, being overconfident in the ability to predict and control the future, or seeking to explain complex issues with single causes (Cork et al., 2023).

While recognizing that narrow and shallow approaches are often well-intentioned and better than doing nothing, it is essential to understand their limitations to avoid getting stuck in a dead end. Building on previous chapters, I try to provide a more complete overview of the limitations and risks of such approaches, to pinpoint the reasons why moving beyond silos and tackling root causes is crucial and should be a priority for coming decades.

5.2.1. Limitations of narrow and shallow approaches

Narrow and shallow – conventional – approaches appear insufficient to halt the biodiversity and climate breakdown, because they tend to lack effectiveness and may generate collateral damages, which may even worsen or sustain over time the targeted problems. Here, I focus only on the consequences stemming specifically from the narrow and shallow characteristics of these approaches. Other features that lead to a lack of effectiveness were presented briefly in section 1.3.1.5.

5.2.1.1. Too slow

By favouring incremental steps, conventional approaches appear too slow to bring about the necessary changes (Wise et al., 2014). Despite decades of sustainability interventions, at current rates, biodiversity loss and climate change risk crossing tipping points, leading to irreversible changes.

5.2.1.2. Spillovers, injustice and pushback

Narrow and shallow interventions can lead to some positive effects for specific outcomes while, at the same time, they may generate significant negative spillovers across scales (social groups, space and time), reinforcing existing problems or inducing new ones (Prudhomme et al., 2020). These ‘off-stage’ (i.e. diffuse, delayed, and distant) burdens, are often not accounted for when designing policy interventions for a given target sector or spatial area (Garrett et al., 2025; Pascual et al., 2017).

As shown in previous chapters, leakage may curtail the intended policy outcomes, such as traceability meant to secure deforestation-free cocoa supply chains, which, by not covering the whole sector, leaves gaps through which non-compliant cocoa can still be sold. In parallel, other land uses not included in the policy may expand

or be displaced in forested areas. Conservation, restoration, and renewable energy expansion all face issues of leakage. Spillovers may also create positive feedback loops (i.e., ‘solutions’ to climate and ecological breakdown participating in worsening it). For instance, climate adaptation actions can increase emissions thereby increasing the need for adaptation (Barnett & O’Neill, 2010) – with a classic example being the increased use of energy-intensive air conditioners in response to the health impacts of heat-waves (Kovats & Hajat, 2008). Rebound effects can also happen, in which initial environmental gains are ultimately offset due to increased production and consumption of ‘green’ products, as was shown with increased efficiency in recycling practices leading to increased consumption of plastics (Chakori et al., 2022), or with agricultural expansion following intensification (Rodríguez García et al., 2020).

Spillovers may also affect a variety of other problems, considered ‘peripheral’ to the problem addressed, which may have been overlooked, especially those that did not (yet) become *policy* problems (Bastos Lima et al., 2019) – i.e. problems recognized by the policy agenda. For instance, sectoral sustainability assessments tend to ignore the land requirements of the assessed intervention, as well as conflicts and competition with other land uses (Meyfroidt et al., 2022). Another example is how REDD+ may favour the commoditization of forest carbon to fit carbon market requirements with little considerations of biodiversity and local communities (Baynes et al., 2021).

The sustainability agenda often fails to consider the socio-economic implications of considered interventions, and can increase social injustices and inequalities (Vatn et al., 2024); e.g. through the establishment of protected areas or large-scale plantations leading to land grabs and population displacements as discussed in chapter 4, or the scaling of technology in agriculture generating exclusion and marginalization of poorer farmers (de Roo et al., 2019). Affected publics may reject the intervention altogether, as was the case with large-scale oppositions to carbon pricing/taxes in Australia, Canada, and France (Patterson, 2023). Thus, by not adequately accounting for these considered ‘peripheral’ problems, beyond their intrinsic importance for social justice, siloed interventions also hamper their own feasibility and potential long-term existence (Kaiser & Patterson, 2025; Pascual et al., 2022). Further, inequalities are increasingly recognized as simultaneously driving and being strengthened by the climate and ecological breakdown – not taking them in account thus ultimately prevents ‘solving the problem’, as further explored below (section 5.3.3).

5.2.1.3. Entrenching incoherencies

Narrow and shallow approaches also generate situations of incoherence, with policies conflicting with one another. This was shown for the climate policyscape in the UK where policies favouring climate mitigation are confronted to many other policies hindering mitigation (Zambianchi & Biedenkopf, 2024). As IPBES notes, *‘removing environmentally harmful subsidies and regulations is as important as increasing funding for conservation’* (Gurung et al., 2025). Many examples exist; transnational companies or governments committing to zero-deforestation, whilst at the same time placing investments in fossil fuels; expanding BECCS on forested land, or on land that would capture more carbon by being restored, or that would produce more energy if used for solar panels, as discussed in chapter 4. A prominent example is the UN Sustainable Development Goals (SDGs) where, for instance, no link was made between industrialization and energy, and only a weak link between energy and climate (Le Blanc, 2015). Incoherence lies also in the dual goal of the SDGs: to stop climate change and biodiversity loss, while sustaining economic growth (cf. section 5.3.1).

5.2.1.4. Path-dependencies and lock-ins

Implemented interventions can create detrimental path-dependencies and lock-ins²⁷, making the problem harder to ‘fix’ (Kim & Anderson, 1998). For instance, in the case of SSIs in chapters 2-3, introducing economic rewards to farmers to influence practices can perversely lock them in the supply chains that drive deforestation and biodiversity loss and maintain them in poverty, or supersede values related to stewardship of nature (Ezzine-de-Blas et al., 2019). Similar concerns have been raised regarding offset schemes such as REDD+, which can perpetuate policy lock-in by incentivizing inaction on fossil fuel reductions (Asiyanbi & Lund, 2020). Another example is how the EUDR and other zero-deforestation commitments may advantage large and vertically integrated agri-food companies, better equipped for tracing and monitoring their supply chains (Folke et al., 2019; Panwar et al., 2023). Smaller players risk being evicted, further increasing market concentration – thus locking-in affected farmers in supply chains with few or no alternative buyers (McDermott et al., 2025). Higher market concentration gives more negotiation power to fewer actors, who may then have the power to undermine zero-deforestation efforts or further reduce farmgate prices, entrenching poverty and dependence (cf. section Concentration of wealth and power).

²⁷²⁷ Path-dependencies and lock-ins relates to how future pathways are contingent on historical pathways and difficult to change (Wise 2014).

5.2.1.5. Falling short of solving underlying problems

Conventional approaches tend to conform to and accommodate existing economic and political structures and goals (Pascual et al., 2022; Pungas, 2023; Vatn et al., 2024), and, in particular, the majority of UN biodiversity and sustainability policies have been found to advocate economic growth (Otero et al., 2020) – which, as developed in the next section, constitute underlying causes of the biodiversity and climate breakdown. As a result, they fail to sufficiently (in time and scale) constrain unsustainable practices and may even strengthen and reproduce detrimental mechanisms as seen above (Leeuwis et al., 2021; Pungas, 2023).

For instance, as shown in chapter 3, measures to reduce deforestation are bound to be insufficient if they do not address the growing demand for forest-risk commodities and global power and wealth imbalances (Carmenta et al., 2023). As discussed in chapter 4, the global sustainability agenda bypasses the question of the reduction of production and consumption volumes. Instead, it searches for ways to increase efficiency, protect and restore ecosystems, or switch energy production to renewables, without seriously considering ways to directly reduce demand at source. Another example from Chakori et al. (2022) is how interventions to reduce food packaging only tackle symptoms of the problem, while a deeper change of the growth-driven globalised food market and time-deprived society would be required to effectively reduce packaging while preventing side effects. Similarly, Leeuwis et al. (2021) showed how interventions in the food system fail to address issues of inequality because they do not challenge but rather align with the current paradigm.

5.2.1.6. Diverting funding and eroding alternatives

Finally, narrow and shallow interventions may divert funding from other alternatives. As an example, the EUDR has further pressured governments and the private sector to invest in cocoa traceability, hence possibly impacting relative government expenditures on food crops and other environmental health and social welfare priorities (McDermott et al., 2025). Importantly, these interventions also contribute to the erosion of more radical solutions (Kim & Anderson, 1998). For instance, greenwashing practices lessen the pressure to implement more fundamental changes (Chakori et al., 2022). Some argue that restoration and conservation interventions may also feed depoliticising ‘degradation narratives’, justifying enclosures and smallholder dispossession, while diverting attention away from the extent and magnitude of the ecological impacts of continued extractivism (Huff & Brock, 2023).

5.2.2. Implications: Breaking silos and diving deeper

As Biely and Chakori (2025), referring to Béné (2022), point out: ‘*Humans are trapped in a vicious cycle where they constantly have to solve the problem created by the solution they have applied to another problem*’. Some argue that this is how humanity has always evolved (DeFries, 2014), however, today’s multiple interacting and accelerating global environmental challenges and crises call for different approaches (Gurung et al., 2025). Because today’s crises are entangled, they cannot be fully understood nor addressed in isolation (Lawrence et al., 2024). A shift in perspective thus appears crucial²⁸; moving from narrow and shallow approaches to adopting a comprehensive, integrated and systemic approach²⁹ (Garrett et al., 2025). Conventional interventions (those deemed relevant locally) need to be complemented by more transformative actions that catalyse deeper changes and address underlying causes, reconfiguring current structures, policies and regulations (Gurung et al., 2025). It first requires to identify and understand what these root causes are.

5.3. Searching for roots of intertwined problems

In its recent report on transformative change, the IPBES defines underlying causes as ‘*deep rooted interconnected social and cultural patterns that shape, influence and reinforce all drivers of biodiversity loss and nature’s decline*’ (Gurung et al., 2025). They caution that the word ‘causes’ should not be interpreted in a linear sense, since ‘*causality in a systems perspective is better understood as multifactorial, circular or spiraling*’ (Reyers et al., 2022). They also distinguish *underlying causes* (or root causes), which lie ‘*beneath the surface*’, from *indirect drivers*, which are more obvious to the origin of observed problems.

Identifying underlying causes is like following a web of threads from the surface to deeper roots, asking at every step ‘*Why is this so?*’. For instance, current zero-deforestation commitments place the onus for change on socially and economically marginalized communities. These approaches often attempt to stop practices considered ‘harmful’ by merely replacing them with ‘Good Agricultural Practices’

²⁸ Noting that achieving greater integration at various levels was already a core concern of the international community at the Rio+20 conference (United Nations, 2012 in Le Blanc, 2015).

²⁹ This approach would not be without trade-offs. Any intervention benefiting people at one level, on one specific aspect, at one particular time, is likely to go at the cost of other people or aspects at the same or other levels, at the same time or in the future (Leeuwis et al., 2021). However, broadening and deepening the issues and policy mix considered has higher chances to reduce collateral damages and to be effective.

(GAP) in order to safeguard forests. Searching for roots would mean to instead ask the question ‘*Why are these practices existing?*’. If one of the answers relates to prevalent economic poverty, searching for roots would require to keep on and ask ‘*Why is there economic poverty?*’³⁰ – eventually bringing forth larger-scale structural conditions driving exclusion, disempowerment and associated aspects of poverty.

Systems thinking³¹, as a school of thinking focusing on studying systems as a whole, can help in identifying underlying causes. Building on that, transition studies³² provide frameworks to understand past transitions and to hypothesize about how future transitions could unfold (Biely & Chakori, 2025). Particularly, a sustainability transitions lens allows to assess persistent climate and ecological breakdown as a harmful effect of a *dominant regime*³³ in need of change, constituted of a complex web of material and immaterial structural factors (e.g., institutions, technologies, markets, infrastructure, culture), and their interrelations with the ecological systems. It allows to identify key lock-ins which render changes of the development path difficult, as well as the related leverage points having the potential to bring about actual change (Meadows, 1999). Here, I do not intend to apply these frameworks as such, but I draw on the related literature.

Global sustainability crises have been noted to share common underlying causes, inseparable from the dominant political, economic, social and cultural regime (Escobar, 2015). Here, I explore three major underlying causes³⁴: (i) capitalism and

³⁰ ‘Poverty’ is a normative concept; there are rising calls to question what constitutes ‘poverty’ and what enhances well-being in rural place-based communities, instead of attempting to ‘resolve’ local economic poverty (Carmenta et al., 2022).

³¹ This approach gives special attention to understanding the structures and interrelations between the parts that comprise the system, and how a system’s behavior arises from these (Senge, 1990 and Sterman, 2000 in Arias-Gaviria et al., 2021).

³² There are many different conceptions (see Biely & Chakori, 2025 for the references): e.g., socio-technical transition theory (Geels, 2002a), socio-ecological transition theory (Gunderson & Holling, 2002), socio-institutional (Loorbach et al., 2017) or socio-economic transition theory (Kemp et al., 2022), transition pathways (Pathways, 2021), the three horizons (Sharpe et al., 2016), the geography of sustainability transitions (GeoST) (Coenen et al., 2012), the sustainability innovation framework (Dabard & Mann, 2023).

³³ According to Geels (2011), regimes refer to ‘*intangible and underlying deep structures (such as engineering beliefs, heuristics, rules of thumb, routines, standardized ways of doing things, policy paradigms, visions, promises, social expectations and norms)*’, while systems captures the ‘*tangible and measurable elements (such as artefacts, market shares, infrastructure, regulations, consumption patterns, public opinion)*’. Hence, a regime is considered to be an ‘*interpretive analytical concept that invites the analyst to investigate what lies underneath the activities of actors who reproduce system elements*’. (Geels, 2011)

³⁴ IPBES identified three key underlying causes, which fall into my two last categories: (i) disconnection from and domination over nature and people; and (ii) the prioritization of short-term, individual and material gains - which are both included in what I call here ‘the dominant worldview’-, and (iii) concentration of power and wealth, which matches with my third category. Hence, IPBES does not explicitly name capitalism in these underlying causes, though the prioritization of short-term, individual and material gains, as well as the two other categories quite match with major characteristics of capitalist systems.

the growth paradigm (section 5.3.4); (ii) the dominant worldview promoting disconnection, domination, materialism and individualism (section 5.3.2); and (iii) the concentration of wealth and power (section 5.3.3.4). These three aspects are not isolated from one another. They interact strongly and have historically co-evolved, supporting each other in ways that maintain the dominant regime in place. I then shortly highlight the role of lock-ins and barriers in preventing change (section 5.3.4).

5.3.1. Capitalism and the growth paradigm

5.3.1.1. Environmental impact of growth

Economic growth is widely recognized as driving the growing consumption and production of goods and services (Wiedmann et al., 2015), which primarily require energy and rely on the exploitation of nature, resulting in multifaceted consequences such as climate change (Schöngart et al., 2025), deforestation (Hoang & Kanemoto, 2021; Pendrill et al., 2019), biodiversity loss (Marques et al., 2019), overfishing (Tickler et al., 2018), fertilizer and pesticide run-off from intensive commodity production (M. Clark et al., 2018), to name just a few. Its effects extend way beyond individual countries' jurisdictional borders through telecouplings (cf. section 1.2) – and across markedly different sociocultural contexts within countries.

Higher levels of consumption and environmental impacts are strongly linked with affluence, itself driven by economic growth (UNEP, 2024). Overall, economies with higher GDP per capita tend to consume more raw materials and energy (UNEP, 2024). High income countries represent one-third of the global population but have material consumption footprints 60% higher than middle-income countries, and thirteen times the level of low-income countries (IRP, 2019). These countries also cause ten times more climate impacts through consumption than low-income countries (IRP, 2024). The world's rich citizens cause higher environmental impacts (Chancel et al., 2022), with the world's top 10% of income earners estimated to be responsible for 25-43% of environmental impact (compared to 3–5% for the world's bottom 10% of income earners) (Teixidó-Figueras et al., 2016).

Quite ironically, the intrinsic logic of escalation of modern societies threatens to undermine their own viability. As per Rosa et al. (2017), '*Stabilization through escalation is the formula for its success as well as for its downfall*'. This is particularly evident with the rising environmental breakdown threatening commodity production, e.g., through crop failures. While it is clear that economic growth is a

major driver of the ecological and climate crises, the search for underlying causes requires to further look into the mechanisms fuelling growth.

5.3.1.2. Mechanisms spurring growth

Although any productivist system may spur growth, today's growth imperatives can be attributed to capitalism as the currently dominant socio-economic system in affluent countries (Wiedmann et al., 2020). Arguing that there is no clear definition of capitalism that encompasses its modern varieties while being circumscribed enough to remain meaningful, Harris and Delanty (2023) proposed a definition based on seven key elements capturing its primary economic logic: (1) the pursuit of profit and its private appropriation, (2) free enterprise and the competitive market, (3) wage labour and the production of commodities, (4) property rights, (5) the financial infrastructure of money and investment that makes possible credit and debt, (6) a highly variable degree of state regulation, and (7) a propensity for growth as the productive re-investment of profit.

Capitalism fundamentally creates the conditions for (e.g., allowing for the private ownership of the means of production and competitive markets) and fosters private profit maximization. These profit-seeking practices lead to aggregate growth dynamics, and growth feeds back into the system (Blauwhof, 2012; Parrique, 2019). Examples of such practices include planned obsolescence, creating the need for replacement; advertising, creating a sense of lack; privatised goods and services with increased access prices (e.g., health insurance, water, energy). Scholars argue that these practices create and maintain forms of *artificial scarcities*, 'artificial' in the sense that they are an outcome of current dominant societal structures while they could be avoided (Hickel, 2020). They spur the need for higher incomes, hence driving growth (Hickel, 2020).

Several mechanisms can explain why capitalist systems rely on growth, which I exemplify here for three main actors following Wiedmann et al. (2020).

- **Firms:** Competition requires private firms to make profit to stay competitive and survive; (i) by reinvesting their profit into more efficient processes, innovation of new products or advertising (Blauwhof, 2012) and (ii) by being able to offer attractive dividends to shareholders to secure financial capitals. Shareholders, via financial markets, are thus crucial in spurring growth, as they provide (interest-bearing) capital and channel it where it is most profitable (Smith, 2010).
- **States:** '*Growth has become the leitmotif of capitalism and also has a quasi-normative value in being an unquestioned axiom that governments must pursue as a good in itself*' (Harris & Delanty, 2023). Two main mechanisms explain why capitalist economies need growth: (ii) to attract and keep financial capital (as,

like firms, states are in competition); (ii) to keep employment constant in the face of rising labour productivity (e.g., through mechanization). Without growth, non-bankable firms would close down, jobs would be lost, and states would lose tax revenues (e.g. from labour and value-added taxes), while social security expenditures would rise, putting social stability at risk (Rosa et al., 2017).

- **Workers:** In capitalism, workers must compete in labour markets to sell their labour power in order to earn a living. This competition can entrench inequalities, create dependencies and strengthen domination patterns, as described in sections 5.3.2 and 5.3.3. It also yields another dynamic directly spurring growth: the *efficiency consumption*. In capitalism, individuals need to manage high workloads while staying competitive. They do this by increasing time and cost efficiency, e.g., by investing in cars, kitchen appliances, computers and smartphones, or by using online trade. It is another facet of the rebound effect, where time-efficiency innovations foster the acceleration of life, ultimately leading to lock-ins, e.g. when owning a car or smartphone becomes a prerequisite for work (Siemoneit, 2019). This often goes hand in hand with the commodification of products and services which used to be provisioned through more time intensive commons or reciprocal social arrangements, e.g. convenience food vs. cooking together (Parrique, 2019). This replacement of human labour with energy- and material intensive industrial production typically increases environmental pressures (Ivanova et al., 2020).

Together, these dynamics create multiple dependencies of workers, firms and states on a well-functioning capitalist system and favourable conditions for capital accumulation through aggregate growth and high consumption (Blauwhof, 2012). Following this analysis, it is not surprising that the growth paradigm is hegemonic, i.e. the perception that economic growth solves all kinds of societal problems, that it equals progress, power and welfare (Kallis et al., 2018). This can explain the political difficulty of addressing detrimental production and consumption patterns. It was observed with global development agendas failing to integrate this key dimension in spite of it being identified since the first Earth Summit as a key ingredient of sustainable development paths (Le Blanc, 2015). Overall, as developed in section 5.2, sustainability policies have almost only focused on improving the efficiency of production practices and sourcing processes; e.g., traceability, ZDCs, GAP, protected areas, renewable energy. There has been comparatively little focus on reducing production and consumption, whether of deforestation-risk commodities (Garrett & Rueda, 2019; Kumeh & Ramcilovic-Suominen, 2023), or of any goods as discussed in chapter 4 (section 4.3.6).

In recent decades, and in line with the narrow and shallow approaches discussed earlier, the ecomodernist³⁵ concept of *green capitalism* or *green growth* has emerged, expressing the belief that the current system and economic growth can be maintained with minimal impact on the environment (i.e. decoupling) – with technology playing a central role in bringing about this change (e.g., Ecomodernist Manifesto, 2015). A major gateway to ensure decoupling is the increase in the efficiency of resource use, as discussed in chapter 4 (section 4.3.5). Relative decoupling, whereby both GDP and resource use grow, but at different rates with GDP growing faster than resource use, has been observed at global and country-level over long periods (Vadén et al., 2020). A number of countries in the global North have recently managed to reduce emissions while still growing their economies, i.e. absolute decoupling (Le Quéré et al., 2019). Yet, sustained and ‘Paris-compliant’ absolute decoupling, whereby consumption-based resource use (i.e. including ‘imported’ emissions) declines at a sufficient rate while GDP grows, has not been observed so far (Haberl et al., 2020; Vogel & Hickel, 2023) – noting in addition that decoupling is often assessed for GHG emissions, while it is also required for other environmental impacts (Parrique et al., 2019).

Ultimately, infinite economic expansion on a finite planet is, presumably, impossible (Haberl et al., 2020). Since reducing consumption and production (and not just ‘greening’ them) is antithetical to capitalist economies, shifting away from this growth-oriented system is a prerequisite for sustainability.

5.3.2. The dominant worldview

The current dominant worldview stems from the Western world and became dominant in the past few centuries through colonialism, imperialism and globalization, and the marginalization of other worldviews (Escobar, 2015). The capitalist regime and global sustainability crises are rooted in this worldview and its associated behavioral and cultural patterns.

Again, it is hard to categorize these strongly interacting and interdependent patterns. Here, I only shortly develop two of them, playing a central role in the climate and ecological breakdown: (i) disconnection with nature and domination over nature and ‘othered’ people, (ii) materialism and individualism.

³⁵ Ecomodernism is ‘an environmental philosophy rooted in the belief that technological progress can allow humans to flourish while minimising our impacts on the environment, in particular by freeing up land for conservation by intensifying the production of food and other resources using technology’ (Breewood et al., 2022).

5.3.2.1. Disconnection and domination

Like many other forms of social organisations, capitalist economic systems are characterized by social hierarchies and relations of domination. To be maintained over time, capitalism require cheap productive labour to maximize profit, ‘natural resources’ to extract, as well as reproductive labour (e.g., unpaid domestic work). Hence, a disconnection from nature and structures of domination, embodied in systemic sexism, racism, classism, and speciesism, play a critical role in perpetuating the dominant growth-oriented regime in place. Capitalist systems and environmental destruction, in turn, reproduce inequalities, social hierarchies and relations of domination (cf. section 5.3.3).

Today, the dominant worldview places humans as separate from and superior to nature, while nature is considered as composed of objects, conceived as ‘natural resources’ that can be exploited without consequences or need for reciprocity (Luyckx, 2020). The process of *disembedding* describes how modern societies have become uprooted from land and nature, resulting in diminishing knowledge of, and attention to, nature’s contributions to people (Borgström Hansson & Wackernagel, 1999). This material, symbolic, emotional and perceived disconnection allows for the domination and destruction of nature (Chagnon et al., 2022). It is particularly pervasive with long-distance environmental problems such as deforestation overseas (Bastos Lima et al., 2021). This disconnect can also be found in the emergence of a ‘repair mode’, in which nature is conceptualized as a balance sheet of substitutable capital, allowing neo-colonial enclosures in the name of fixing a broken world (Huff & Brock, 2023). Similarly, monofunctional landscapes were found to reinforce the growing disconnect between people and the land (Fischer et al., 2017). These landscapes optimized for crop production involve few people in their management, with the benefits flowing to a small set of actors, while local people tend to import food (Fischer et al., 2017). This leads to lost ways of knowing, or the ‘*homogenization of the mind*’ (Maffi & Woodley, 2012).

The emergence and history of this human-nature divide in Europe can be traced back to Greek philosophers, or the Cartesian dualism³⁶. Scholars however argue that the 18th century and the Enlightenment played a pivotal role therein and engaged

³⁶ Descartes introduced the mechanistic understanding of the physical world, considered purely as inert matter. It is argued that this view has contributed significantly to an instrumental and disenchanting conception of the physical world, reduced to mere resources for human consumption (Kureethadam, 2017). He also supported the idea of a metaphysical dualism: the *res cogitans* (i.e. humans) and the *res extensa* (i.e. the rest of the physical world, including non-human beings), both being diametrically opposed, and all rational and intellectual properties possessed solely by humans. Such an ontological divide between humans and non-humans is argued to be at the very basis of the domination and exploitation of the natural world by humans (Kureethadam, 2017).

the development of a model of society ‘*based on instrumental rationality, the imperative of efficiency, the scientific objectification of the world and the fragmentation of knowledge, while neglecting, in public affairs, anything relating to affect, the holistic and sensitive understanding of reality and its symbolic, cosmic and poetic dimensions*’ (Luyckx, 2020, translated from French). This paved the way to the idealization and promotion of control-oriented, mechanistic and reductionist ways of reasoning (Stirling, 2019), considered as rational. In the same movement, it participated in the devaluation of the realm of emotions – considered as irrational and impeding practical rationality (Simpson, 2014), thus contributing to the marginalization of care and empathy. Scholars working on emotional intelligence argue that a lack of empathy and care prevents ethical reasoning (Warren, 2015).

Further, it is argued that this human-nature framing also stems from patriarchy³⁷, whereby certain types of masculinity are privileged, mostly those that ‘*erase emotional connection, suppress empathy, promote detachment, and prime men for violence*’ (de Sousa Santos 2018 in Bossert et al., 2024). By praising these norms, patriarchy participates in reinforcing rational thinking, in driving (emotional) disconnection from nature, and cultivating a culture of domination, facilitating the exploitation and destruction of nature (Bossert et al., 2024).

Importantly, a whole body of research shows how the objectivization of nature also insidiously affects how specific groups of people are perceived. It is argued that those deemed part of, or closer to, nature are being devalued, and considered sacrificial, vulnerable, or in need of protection, development, or change – namely women, Indigenous Peoples, elderly, people living with disabilities and racialized peoples (Frantzeskaki et al., 2025; Warren, 2015). Hence, the disconnection and domination of nature constructs and reinforces racial, gendered and anthropocentric hierarchies.

Domination patterns and the dominant worldview also participate in excluding and marginalizing other ways of knowing and living with nature (Arora & Stirling, 2023), though they hold key potential in inspiring other ways of relating. They further tend to place the blame for environmental destruction, such as deforestation, on farmers and local communities (considered as less capable of environmental stewardship), diverting attention from underlying causes such as the growing demand for land, rooted in the globalized, growth-oriented capitalist economy (Carmenta et al., 2023).

³⁷ Bossert et al. (2024) define patriarchy as ‘*the nexus of social, political, and economic processes, norms, and structures that produce and sustain male supremacy, male authority, and gender inequality*’.

5.3.2.2. Materialism and individualism

The dominant capitalist regime defines societal progress as dependent on continuous and limitless economic growth and wealth accumulation. This *One World development model* (Escobar, 2015) propagates cultural and social norms according to which individual material wealth accumulation (extending far beyond the satisfaction of basic needs) equates to success in life, happiness or wellbeing (Cabanas & Illouz, 2018). This materialist norm stimulates overproduction and overconsumption (Ivanova et al., 2020; Wilk, 2022).

The concept of *meritocracy* plays a central role in allowing for individual wealth accumulation, and in protecting and legitimizing the associated interests of the wealthy (Harris & Delanty, 2023). Further, coupled with the view of humans as self-interested, it promotes prevailing governance patterns based on competition rather than cooperation (Kothari et al., 2019 in Carmenta et al., 2023).

Prevailing competition, and the prioritization of short-term, individual and material gains, has come at the expense of values of community, care, solidarity and the maintenance of the '*social and ecological fabric*' (Gurung et al., 2025). This view has, for example, justified the privatization of public services (Brogaard & Helby Petersen, 2022), while it also places the onus on individual responsibility instead of looking into systemic mechanisms; e.g., in terms of exposure to risk (e.g., discourses of youth responsabilisation in juvenile justice services (Liebenberg et al., 2015)), or climate action (e.g., prioritization of individual behaviour change instead of collective action for system change (Shove, 2010)).

5.3.3. Concentration of wealth and power

Wealth is being increasingly concentrated to the benefit of a few. Wealth inequality within countries is at an historic high, while wealth inequality between countries remains high despite gains made by some in recent decades (Chancel et al., 2022). Noticeably, there is a growing gap between increasing privately-held wealth and decreasing public wealth in rich countries (Chancel et al., 2022). Historical patterns of wealth appropriation between the Global North and the Global South, rooted in colonialism, continue today (Sullivan & Hickel, 2023).

Concentration of wealth and power constitutes an underlying cause of the ecological and climate breakdown through multiple mechanisms: (i) affluent and powerful actors benefit from greater political influence (i.e. power) keeping the growth-driven regime in place and (ii) they have greater social influence driving consumption norms across the population; (iii) the resulting inequalities entrench poverty

and dependence, leading precarized and marginalized people and nations to work for the system.

5.3.3.1. Political influence

Wealth concentration translates into greater power for the wealthy, who benefit from preferential access to decision-making processes and greater means of political influence (Green & Healy, 2022). Individuals, corporations and other organizations that benefit from current systems often wield their power and resources to protect their interests (Brulle & Downie, 2022; Franta, 2022) – which most often favour the status quo or increased wealth concentration, supporting the entrenchment of the growth-driven capitalist system (Green & Healy, 2022). They are able to support and ‘ossify’ particular narratives and framings, while marginalizing alternative narratives (Brockhaus et al., 2021; Leach et al., 2010). Research has found an increasing dominance of neoliberalism (Gareau & Lucier, 2018) and pro-growth ideologies (Manzano et al., 2016) in global governance institutions. Hence, critics posit that these institutions, determining national economic policies and regulations, are designed to protect rather than challenge the dominant regime (Hickel et al., 2022).

In the context of land systems, powerful private actors may promote their business interests ahead of sustainability or resist regulatory efforts, as mentioned in section 1.3.1.5. Chapter 3 pointed out that the cocoa sector is riddled with vested interests from actors in government, companies and some civil society organisations (Amuzu et al., 2022; Odijie, 2018). This participates in fuelling the concentration of investments into the cocoa sector, as opposed to other sources of local livelihoods (McDermott et al., 2025).

In contrast, marginalized people have less access to decision-making (J. K. Clark, 2018), lowering their ability to protect their rights. This is particularly striking in the context of frontiers where forest-based smallholders are displaced by the expansion of large-scale cattle ranches and commercial crops (le Polain de Waroux, 2025).

Further, powerful affluent actors can also target social movements and individuals that speak out for change, and undermine efforts by civil society. Indigenous Peoples and local communities, as well as citizen groups and social movements, often face discreditation and even direct physical violence for resisting the drivers of destruction (Global Witness, 2022; IPES-Food, 2024).

5.3.3.2. Social influence

The way of life of the wealthy is often perceived as an aspirational goal. This fosters the pursuit of *positional consumption* whereby every actor strives to increase their position relative to their peers. Once basic material needs are satisfied, the proportion of purchase of positional goods – i.e. signifying social status, and tending to be expensive – increases (Frank, 2001). Status indeed matters, with relative income being one of the strongest determinants of individual happiness (A. E. Clark, 2018). This resembles an endless pursuit of happiness, as the average consumption level rises and thus even more expensive positional goods become necessary, while in the aggregate societal wellbeing stagnates (A. E. Clark, 2018). Hence, inequalities spur growth by contributing to an increase in consumption overall.

5.3.3.3. Entrenching poverty

Wealth and power concentration generate inequalities, and inequalities perpetuate inequalities. This reflects what in system terms are called *success to the successful* feedback loops, which are positive feedback loops that reward the successful with the means to succeed (Meadows et al., 1972). The concept of *coloniality* reflects how appropriation and extraction of resources both fuels and reproduces the wealth, power and privilege of specific groups (Quijano, 2007), resulting in relations of domination, both of people over nature and people over other people. Historical patterns of wealth appropriation, entangled with colonialism, is one of the key factors maintaining substantial parts of the global population in a state of poverty (Sullivan & Hickel, 2023). Many examples exist of such ‘self’-perpetuating inequalities leading to *poverty traps* (Haider et al., 2018) and preventing change, all highly associated to unequal political power, as exposed above. Some were discussed in chapter 3 – I further develop some here.

- ***Unequal value distribution along supply chains.*** In the context of forest-risk commodities, relatively few highly capitalized firms capture large shares of the profits, while smallholder farmers struggle to earn a living income and are vulnerable to price fluctuations (van Vliet et al., 2021; Waarts et al., 2021). The majority of cocoa produced in Ghana and Côte d’Ivoire is exported unprocessed to international markets, mainly Europe. Meanwhile, the collective fortunes of the Ferrero and Mars families surged to over \$160 billion in 2023, more than the combined GDPs of Ghana and Côte d’Ivoire (Oxfam, 2024).
- ***Unequal trade agreements and exchanges.*** Substantial research finds that trade-based policies risk entrenching inequalities in global trade itself (Kleinschmit et al., 2023). Capitalist and neo-liberal arrangements put in direct competition actors with very unequal means, e.g., Southern producers facing subsidized large-scale

agriculture of Western countries and being driven out of business as a consequence (Garrett & Rueda, 2019). In addition, unequal exchanges allow wealthy countries to displace the social and environmental impacts of their consumption in poorer regions or the world, causing biodiversity loss, land degradation, water pollution, among others, as well as unhealthy and dangerous working conditions for affected populations (Givens et al., 2019).

- **Unfair financial debts.** Many countries of the Global South are under constant pressure to service foreign debts, some stemming from colonial times – hence questioning their legitimacy, and some of which have a principal that has been repaid many times over (Hickel et al., 2022). Financial markets tend to place greater trust in wealthy countries, granting them loans at more favourable interest rates, which allows these nations to ‘roll-over’ their debt at a relatively low cost, while poorer countries see their debt burden (i.e. the yearly cost of their debt) being much higher. As a result, and following Structural Adjustment Plans³⁸, low-income indebted countries are constrained to increase export earnings by selling out (‘cheapening’ (Patel & Moore, 2017)) their natural and human wealth, to the detriment of their social, economic and environmental integrity (Kumeh & Ramcilovic-Suominen, 2023). This constitutes a massive economic drain from the Global South to Global North (Hickel et al., 2021). Cocoa, for instance, plays a crucial role in servicing Ghana’s rising international debt (Asante-Poku & van Huellen, 2021). In 2023, as their interest costs climbed to a 20-year high, ‘developing’ countries spent a record \$1.4 trillion to service their foreign debt (World Bank, 2024). Both Côte d’Ivoire and Ghana are currently in debt crisis³⁹, where *‘debt payments are undermining a country’s economy and/or the ability of its government to protect the basic economic and social rights of its citizens, for example by providing access to healthcare, education and social protections’* (Debt justice, 2024).
- **Unequal opportunities.** Overall, wealthier and powerful actors have greater means, hence greater opportunities – thus perpetuating their dominant position. In the context of land use frontiers, powerful and capitalised firms are able to identify and capture ‘abnormally’ high land rents compared to other actors. Their greater access to technological, financial, political and informational means enable them to find specific areas valuable and worth exploiting (Meyfroidt et al., 2024). Another example is how wealthy countries and individuals are better placed to face

³⁸ In the 1980s, the International Monetary Fund and World Bank provided structural adjustment loans which required indebted nations to institute a variety of economic policy reforms, including devaluing currency, reducing government spending, liberalizing trade, and privatizing government assets. The underlying logic attempts to stimulate economic growth to generate hard currency for debt repayment by increasing exports and decreasing spending (Shandra et al., 2008).

³⁹ Defined as government external debt payments greater than 15% of government revenue, and a large financial imbalance with the rest of the world; either a net international investment position of -30% of GDP or worse, or a current account deficit averaging over 3% per year for three years (Debt justice, 2024).

the consequences of nature's destruction (Christensen et al., 2023), while indebted low-income countries and people lack means to implement initiatives that would make them more resilient (Tubiana & Fox, 2023).

These inequalities within and between countries intersect with inequities across race, gender, age, disability, place, or class, thus reinforcing patterns of domination among people (Malin & Ryder, 2018) (cf. section 5.3.2.1).

Ultimately, by keeping people and nations in poverty, these inequalities generate multiple dynamics participating in the ecological and climate breakdown: (i) they reduce people's and nations' capacity and resources for sustainable land use and sustainability in general; (ii) they constrain people and nations to participate in the system (e.g., accept a low-pay job for a harmful corporation, sell out resources at low-price). This in turn perpetuates the dominant regime which degrades ecosystems and contributes to climate change.

5.3.4. Lock-ins and barriers

Multiple intertwined lock-ins and barriers, which include the underlying causes presented above, maintain the dominant regime in place and make it resistant to change. The IPBES report on transformative change dedicates an entire chapter to it⁴⁰. Let us just go over three of them:

- ***Inadequate political systems.*** Beyond being influenced by vested interests from powerful actors, political systems are riddled by short-term thinking, the lack of political will, or the fear of losing out, to name a few (Frantzeskaki et al., 2025). The persistence of socially and environmentally harmful subsidies exemplifies this trend (Sumaila et al., 2019).
- ***The difficulty of changing perspective.*** Beyond institutional and psychological mechanisms, this can stem from a lack of diversity, particularly in decision-making arenas, which may create ontological and epistemological lock-ins, where

⁴⁰ They are organised in five challenges, each comprising specific barriers: **(1) Persistent relations of domination particularly those emerged and were propagated in colonial eras** (Categorizing and ranking; Measuring, simplifying and managing; Justifying resource concentration); **(2) Economic and political inequalities** (Wealth and power shaping policy; International monetary system constraining policy autonomy; Shareholder profit overriding public interest; Perceived risks of transformative change); **(3) Inadequate policies and unfit institutions** (Rigid political/administrative boundaries; Uncoordinated institutions; Social and ecological cycle mismatches; Understanding and measuring problems of fit); **(4) Unsustainable consumption and production patterns and individual habits and practices** (The ideology of perpetual growth; Socio-economic disparities; Telecouplings; Ingrained practices and habits); **(5) Inaccessible clean technologies and uncoordinated knowledge and innovation systems** (Poor availability of sustainable technologies; Ineffective coordination of knowledge and innovation systems; Inappropriate knowledge transfer and access to sustainable innovations; Unregulated emergent technologies; Marginalization of Indigenous and local knowledges) (Frantzeskaki et al., 2025).

people get trapped in an echo chamber (Unerman, 2020). An example is how anomalies in science tend to be interpreted as to fit the paradigm, rather than questioning the paradigm (Kuhn & Hacking, 2012).

- ***Felt precarity and fear of losing.*** This limits people's capacity to mobilize for, or may even foster their opposition to, transformative change (Green & Healy, 2022).

Together, these lock-in and barriers pose significant challenges to advancing sustainability agendas. Transformative change operates not only by fostering desired pathways but also by weakening the forces that resist change (Martin et al., 2020).

5.4. Tackling root causes

Tackling major root causes of the ecological and climate breakdown entails profound transformation of the dominant regime, which require (i) eliminating the need for growth, (ii) shrinking the detrimental aspects of the dominant worldview and growing other, beneficial, worldviews, (iii) stopping and reversing wealth and power concentration. It calls for broad and deep changes to transform views, practices and structures (Gurung et al., 2025). The complexity is that tackling these root causes both *requires* and *allows for* fundamental changes. It needs simultaneous transformation of the way we *do* and the way we *think*, as both feed into the other. To say this is a challenge is an understatement. However, it is already existing and happening, in many places, though obscured by the dominant regime.

I do not have the pretention to cover here all required actions and changes which would address root causes of today's crises. I picked here what speaks to me the most, and which I feel goes perhaps a step further than more conventional calls. I first explore alternative economic models, focusing on degrowth and, more specifically, its aspects which in my view more directly tackle the roots of the growth paradigm. Then I further expand on the need to build societies rooted in care. By doing so, and looping back to the beginning of the chapter, I hope to help taking down some blinders (starting by my own) and participate in fostering discussions about the many possible alternative aspirational futures.

5.4.1. Navigating thresholds, a degrowth perspective

Diverse alternative economic models to the dominant growth-driven capitalist system are increasingly being discussed (and have been for a long time). They differ from the ecomodernism approach in how they evaluate the possibility of decoupling; some are 'agnostic', while others affirm that it is most likely impossible to

absolutely decouple GDP growth from environmental impacts. Recent accounts summarize their contours and proposals (see e.g., Ghosh and Pearson, 2025; Kallis et al., 2025; Wiedmann et al., 2020).

Overall, these alternative models intend to balance producing sufficiently to fulfil societal needs on the one hand, while remaining within biosphere's limits on the other hand – an aim captured by the concept of *sufficiency*, defined by IPCC as '*a set of measures and daily practices that avoid demand for energy, materials, land and water while delivering human wellbeing for all within planetary boundaries*' (IPCC, 2023a). Similarly, *sufficiency corridors* (or fair consumption space) are comprised between minimum thresholds of production to meet basic needs, and maximum limits to prevent ecological overshoot (Bärnthaler, 2024; Fuchs et al., 2021), which depend on local contexts (Erb et al., 2024). For impoverished communities this would entail addressing socially unsustainable underconsumption to achieve a more equal distribution of wealth, overcome poverty, and guarantee a minimum level of prosperity (Bärnthaler, 2024; Fuchs et al., 2021).

Some models are considered as more reformist and others as more radical. For instance, a-growth (Van Den Bergh, 2017), steady-state economics (Daly, 2014), and doughnut economics (Raworth, 2017), aim to achieve the required socio-ecological transformation by reforming today's dominant institutions to become independent from GDP growth, which they believe is possible in current capitalist economies and democratic states. Conversely, more radical approaches, such as degrowth, argue that the needed socio-ecological transformation will necessarily entail a shift beyond capitalism or current centralised states. In any case, there are strong overlaps between these models, with many similar policy proposals⁴¹ (Wiedmann et al., 2020).

Overall, this chapter aims at identifying and discussing ways of tackling root causes of multiple intertwined problems. Markedly, degrowth aims at tackling major 'pillars' of capitalism, i.e., private property, profit, wage labour, and money, which could be seen as deeper roots of today's problems. Hence, here I focus on degrowth, which has gained traction in recent years. A proposed definition of degrowth is the '*downscaling of production and consumption to reduce ecological footprints, planned democratically in a way that is equitable while securing*

⁴¹ Examples of similar advocated policies include stringent eco-taxes or cap-and trade systems, directed investments in sustainable industries and public institutions, wealth redistribution through taxation and a maximum income, a guaranteed basic income and/or reduced working hours (Wiedman 42,77), or replacing (or complementing) GDP by indicators of social and ecological health.

wellbeing' (Parrique, 2019). The reduction of GDP is not an aim *per se*, but rather a likely outcome of the necessary changes (D'Alessandro et al., 2020).

I only discuss here some of what I consider major salient points of the degrowth agenda, though many other proposals exist (see Fitzpatrick et al., 2022).

5.4.1.1. Creating sufficiency for all

In contrast to the *artificial scarcities* generated by growth-driven systems (cf. section 5.3.1.2), degrowth is presented as a way of creating sufficiency for all, by creating the conditions for people's living needs to be fulfilled, hence freeing them from the need to work intensively to earn more (Hickel, 2020). I present here how degrowth seeks to achieve sufficiency for all, organised into five main categories of complementary (and overlapping) approaches.

1. Stop stimulating the urge for more; e.g., by limiting or banning advertisement, criminalising obsolescence, guaranteeing the right to repair, and by promoting lifestyles of sufficiency. The latter would entail discouraging luxury consumption (e.g., through quotas, progressive taxes on consumption, excise taxes on luxury goods) and encouraging voluntary simplicity (e.g., through bike infrastructures, collective housing, shared utilities, repair cafés, decommodified hobbies) (Fitzpatrick et al., 2022).

The following points would also limit positional and efficiency consumption, by addressing inequalities and competition between wage labour workers.

2. Prevent wealth inequalities through financial tools, with, on the one hand, measures limiting the possibility of accumulating wealth, e.g., maximum wages, highly progressive income taxes, taxes on inheritance and wealth (e.g., maximum authorized wealth), and, on the other hand, measures directing wealth towards poorer people, e.g., minimum living wages, strong social security systems, universal basic income schemes, reparations for ecological debt (Fitzpatrick et al., 2022).

3. Address private property and associated hierarchical structures. The private property of the means of production puts managers and shareholders in a position of power, where they can take decisions which may go against ecological and human needs by seeking profit maximisation (e.g., pull down labour wages, increase prices, use polluting practices), while being a key feature of wealth capture (Borrits, 2018). Hence, building democratic, not-for-profit, and collectively owned business models appears central, such as cooperatives, and smaller horizontal or self-managed enterprises (Democratizing Work, 2020; Fitzpatrick et al., 2022; Parrique, 2019) – which goes hand in hand with relocating activities, also

promoted as a way to cut down GHG emissions while fostering local resilience (Fitzpatrick et al., 2022).

In the same line of thought, defending and reclaiming the commons against their privatisation is a key part of the equation. This means switching to the local, democratic ownership of essential infrastructure such as banking⁴², energy, education, healthcare, local government, telecommunications, transport, waste, and water. The promotion of social, collective and cooperative housing is also seen as key to reduce rents (combined with rent caps and controls, progressive property taxes, etc.) (Fitzpatrick et al., 2022; Parrique, 2019).

Collectivising⁴³ businesses and essential services means shifting decision-making power from corporate managers and shareholders to workers and local communities. It is viewed as central to ensuring access to essential goods and services to all⁴⁴. Collective decision-making, e.g., involving workers and external stakeholders, is expected to decrease environmental pressures and favour higher incomes for workers, and cheaper (or free) access for users, hence reducing the need to work more (noting that these enterprises would also be limited and oriented by legislations). It would also contribute in deflating capital accumulation in the hands of a minority (hence reducing the two underlying drivers of growth and power concentration).

Further, an economy based on collectivised services holds a revolutionary potential hinging on how money could substantially be side-lined. As Parrique (2019, p.437) points out, *‘if I am directly provided access to healthy food, doctors and dentists, and a house, I do not need a cash stipend to buy food, pay doctors and dentists, and rent a house.’* Most welfare states’ social security systems today

⁴² Particularly, (re)claiming banking systems is key to enable transformative change, when considering the primary role of finance in driving growth, by pushing people to work more to pay back debts with interests, but also by lending capitals to and investing in destructive, bankable sectors, and by playing a central role in wealth accumulation (e.g., through speculation). Degrowth proposals focus on (i) neutralising current predatory, profit-seeking finance activities, e.g., by separating traditional banks from investment banks, nationalising monetary creation, taxing financial transactions, closing tax havens, and dismantling banking/financial institutions into smaller, local, and more democratic entities, and (ii) promoting ethical and non-speculative finance, e.g., by creating local and regional currencies, time banks, reciprocity networks and trading systems, self-managed credit unions, cooperative banks, public debt-free money, divestment, and ethical investments (Fitzpatrick et al., 2022).

⁴³ The implication of the state varies in degrowth literature but tends to be limited to a support role, unlike eco-socialism which hinges more on the state, including for the planning of investments and employment, price controls and public ownership of at least the most central means of production to plan their downscaling in a socially sustainable way (Wiedmann et al., 2020)

⁴⁴ Some proposals go further by advocating for a universal provision of basic public services (e.g. electricity, housing, public infrastructure, state services, free consumption goods).

depend on market economies and growth (cf. section 5.3.1.2). In a post-growth economy, solidarity systems would take central stage, with people investing time in services in exchange of access to that or other services. Such systems avoid the ‘detour’ by market economies⁴⁵, hence avoiding to fuel and depend on growth, while building resilience both at individual and societal levels. This social and solidary economy builds on the manifold forms of popular economy existing among communities (e.g., cooperative, associational, mutualistic, redistributive). Inspired by Polanyi’s thought, the economy would be (re)embedded in society by recreating economic systems on the basis of communal dynamics and needs (Escobar, 2015).

4. Close down detrimental companies while ensuring that people have access to fulfilling, purpose-driven work. Work-time reductions is one of the most frequently discussed tools to reconceptualising work, moving in the direction of deprioritising wage labour in society. Part-time jobs allow sharing paid work over more people, hence reducing unemployment⁴⁶. Importantly, it gives people more time for leisure and care, and for getting involved in the collectivized services described above (e.g., food cooperatives, social childcare), as well as in the time-demanding decision processes pertaining to a more horizontal and democratic society. Though the goal would be to maintain living wages (thanks to the redistribution of profits to workers), working less could still reduce higher income and constrain income increases. However, in a society where essential goods and services are either free or very accessible, people’s quality of life would not depend on income (Fitzpatrick et al., 2022; Parrique, 2019).

5. Enabling sufficiency in low- and middle-income countries. Lifting pervasive lock-ins keeping impoverished regions in poverty is key, and requires balancing power relations between nations. A strong lever would be cancelling unfair and unpayable debts of low- and middle-income countries which, as pointed out in section 5.3.3.3, play a central role in perpetuating inequalities. Reforming trade agreements and democratizing international organisations, such as World Trade Organisation, International Monetary Fund, and World Bank, would also be crucial to curb unequal exchange in international trade and create conditions for

⁴⁵ Mastini (2019), cited by Parrique (2019), captures this point as follows: ‘*Financing the welfare state through income taxation is like producing electricity through waste incineration. One kilowatt of electricity produced from an incinerator dissipates more energy than one from a power station because of the two steps involved: first producing the object that becomes waste and then producing electricity from that waste. Similarly funding the welfare state with money coming from income taxation is dissipative because it entails first creating wealth in the market economy and then channelling some of it into public services.*’

⁴⁶ Proposals also include the introduction of job guarantees with living wages.

productive capacity to be reoriented towards achieving social objectives (Fitzpatrick et al., 2022).

To summarize, it is thus by decreasing the need and drive to earn more money, by creating sufficiency for all and eliminating artificial scarcity, that degrowth seeks to decrease production and consumption levels – lifting pressures on nature and climate while ensuring equity and well-being for all.

5.4.1.2. Rebooting democracy

Involving people in political life is a prerequisite for transformative changes. On the one hand, from a practical point of view, all of the above involves many choices to be made, including sensitive decisions such as which production and sectors to scale down or close, or what are the acceptable levels of income inequality. To ensure their social legitimacy, these decisions need to be collectively made. In addition, as outlined earlier (section 5.3.4), current democratic institutions face built-in limitations and would not be fit to the task; they are focused on short-term, politically popular agendas, and, most importantly, they are influenced by wealthy elites. New approaches to governance are thus required (Wise et al., 2014). On the other hand, from a social and psychological perspective, being able to participate in decisions about how society is organised holds great potential for developing a sense of belonging and empowering individuals – which is essential for people to relate and engage, and contribute to changing the dominant regime.

Degrowth, like many other movements, calls for a radical ecological democracy which would give everyone the right and opportunity to participate in decision-making at multiple levels, with decisions grounded in ecological reality. Various mechanisms exist, such as deliberative forums, participatory budgeting (e.g., local communities collectively deciding how to allocate their yearly budget), or local direct democracy (Fitzpatrick et al., 2022). Models of direct democracy, for instance, imply that people, organized in local self-governing assemblies, are directly responsible for making policy decisions (Van Outryve d’Ydewalle, 2024). When required, assemblies choose and mandate members to participate in higher-level councils. These members are ‘*mandated, recallable, and responsible to the assemblies*’ – by doing so ‘*power thus flows from the bottom up instead of from the top*’ (Popp-Madsen, 2017 citing Bookchin). This has been experienced at various scales in many parts of the world for centuries, and currently exists for instance in Chiapas with Zapatista, or in the Kurdistan Rojava.

5.4.2. Societies of care

One could expect that societies based on the degrowth principles outlined above would drastically modify today's dominant values (e.g., materialism, individualism) and relations of domination by building a more equitable system and fostering collaboration. Yet, if only focusing on the reduction of throughput, technically, a society could stay within planetary limits while perpetuating or even worsening gendered and racial discriminations. Degrowth critics point to a trade-off between sufficiency and autonomy, whereby a potentially lower purchasing power, or higher dependence on local services and assets, could reduce the ability to escape situations of oppression (Parrique, 2019, p.442). Moreover, as care work is currently strongly gendered, there is a risk that the burden of new work patterns falls disproportionately on women (Parrique, 2019, p.440). Critics of direct democracy often stress the risk of 'elite' capture, e.g., more self-confident persons taking the lead, with domination patterns coming into play.

As developed in section 5.3.2.1, relations of domination play a crucial role as both causes and consequences of the dominant regime. In short: everything is linked. Hence, beyond their own intrinsic importance, fighting domination relations is essential to enable the emergence of post-growth, or post-capitalist, societies. In other words, an anthropocentric, patriarchal and racist society, in which dominant norms cast emotions, devalue care, and associate success with domination, does not hold much potential for transformative change (or at least, not for the common good). This means that as much as the '*What?*' of transformative change is important, the '*How?*' is decisive; *how* are we building what we wish to build. The process shapes the outcome. In order to avoid trying to solve problems with more of the same, alternative approaches are needed.

Many Indigenous, relational and spiritual worldviews are fundamentally at odds with the dominant worldview, by emphasizing interdependence, unity, holism and relations of care (Atwood et al., 2024). Moving away from disconnection and domination requires our '*social relationships [to] shift towards embodying relations and ethics of care*' (Gurung et al., 2025). Care refers to practices (i.e. it is not limited to wishful thinking, and takes concrete actions: *taking care* of others) prioritizing connections, with an egalitarian commitment. These 'others' are meant to also include ecological processes and all beings in nature (Arora et al., 2020).

Re-connecting people with the land and nature, re-embedding uprooted societies, calls for *relocalization*, based on decentralized, self-managed and grassroots systems, focused on the preservation of soils and ecological integrity (Shiva, 2008). This movement is already happening – for example with farmers re-embedding

agri-food systems in local contexts, emphasizing non-capitalist forms of production and exchange (Garrett & Rueda, 2019). These new circuits advocate for *scaling deep*, intensifying relationships between consumers, producers – people – and the land (Moore et al., 2015). Calls for ‘multifunctional’ landscapes, integrating agriculture and conservation, go in the same direction. Benefits of diverse landscapes are usually experienced more locally, and local people are more likely to be in charge of the management of their land, as compared to monofunctional landscapes benefiting and involving a small set of actors (Fischer et al., 2017). Similarly, degrowth calls for giving constitutional rights to nature, and for shifting values to ecocentric worldviews, abandoning the human-nature divide (Fitzpatrick et al., 2022).

Drawing on ecofeminist, post-colonial and post-development movements and critiques, the struggle against gendered, racial, and anthropocentric domination occupies an increasingly important place in degrowth studies, constantly adjusting to critics (Nirmal & Rocheleau, 2019). Ultimately, degrowth is not only an alternative economic model; it offers avenues for alternative ways of being and relating (*‘faire sociétés’*) – and care should be at the centre.

5.4.3. Implications

5.4.3.1. For cocoa farmers and producing countries

Throughout this thesis, the need to reduce production and consumption of land-demanding products has been put forward, which also applies to cocoa, thus having implications on cocoa farmers and cocoa-dependent economies such as Ghana and Côte d’Ivoire. How a degrowth perspective would translate concretely for commodity dependent producers is a complex question, which calls for democratic discussions. Cocoa remains an economically very important crop for these economies, and particularly for cocoa growing regions. However, as stated in chapter 3 (section 3.4.3), the strong reliance on cocoa translates in dependencies on a globalised economy which, especially in the wake of a global polycrisis, might be problematic (Asante-Poku & van Huellen, 2021). Further, scholars found that policies in the cocoa sector have, historically, created more benefits to political and wealthy elite (Teye & Nikoi, 2023). Whether farmers would be better off without cocoa or not is a tricky question, with a large body of research exploring the impacts of international trade on incomes and food availability and access (Enthoven & Van den Broeck, 2021; Feyaerts et al., 2020), and how this would evolve in the context of climate change (Janssens et al., 2022). Research however is often biased towards income analysis, leaving other aspects of poverty or prosperity unassessed.

Post-development scholarship warns degrowth scholars to ‘*resist falling into the trap of thinking that while the North needs to degrow, the South needs ‘development’*’ (Escobar, 2015). It calls people in the South to avoid endorsing growth as the basis for social improvements; a key criteria being that growth and the economy should be subordinated to *Buen Vivir* – defined as a holistic, de-economized view of social life – and the rights of nature (Escobar, 2015).

Along with calls for relocalized economies of many agroecological peasant movements such as Via Campesina, switching away from the dependency on export-dedicated cash-crops would enable diversification towards food production for local consumption, and leave room to reinforce social and solidarity economy systems. As some countries may not have sufficient land available to produce staple food for self-sufficiency (Franks et al., 2020), and considering potential crop failures due to climate change, cooperation between nations remains essential, but would require power relations to be balanced and production to be *convivial* (Illich, 1973) and empowering for producers, while remaining within planetary boundaries. A first necessary step would be to release the financial pressure, as it is a prerequisite for the freedom to choose and self-defined futures. This links back with cancelling unfair and unpayable debts, and reforming trade agreements and international organisations (cf. point 5 in section 5.4.1.1), also calling for structural changes in finance and stock markets to counteract the financialization process of agricultural products, and counterbalance the price-setting power of ‘grinder-traders’ (Staritz et al., 2023). It would largely participate in creating the necessary conditions for sustainability and sufficiency in cocoa producing countries.

Another important aspect of guaranteeing the necessary conditions for sustainability and sufficiency in cocoa producing regions is to secure essential needs, shifting the focus on reinforcing social security systems, health infrastructures, and, more generally, sustainable community health (Mpofu, 2020). Again, these questions need to be opened and discussed democratically. In this view, CREMAs, which are community-based organisations for nature management building on existing traditional social structures, could be strengthened to support grassroots democracy.

Although insufficient on their own to bring about systemic change, traceability and transparency – the focus of chapters 2 and 3 – are powerful tools for assessing today’s impacts of commodity supply chains and for holding multinationals and central states accountable. They will certainly continue to play an important role in post-capitalist societies, albeit less pivotal, as self-managed not-for-profit enterprises would trigger less social and environmental issues.

For cocoa consuming countries, such a system change would imply relocalized and re-embedded agriculture, with – probably – local delicacies taking precedence and tropical products being consumed sparingly.

5.4.3.2. For the global sustainability agenda

Linking back to the context of chapter 4, aspects discussed in this section 5.4 such as sufficiency corridors and degrowth economies are expected to substantially reduce global land use demands, hence decreasing potential trade-offs and conflicts arising from the global land squeeze. Overall, to move towards that direction, the three major conventions shaping the global sustainability agenda, UNFCCC, CBD and UNCCD, and their related institutions, should embrace more systemic approaches and directly support ways of reducing consumption and production levels, beyond efficiency. IPBES has opened the way with its report on transformative change, and IPCC has for the first time included sufficiency in its sixth assessment report. Without re-opening the points discussed earlier, it is clear that global institutions can and must be transformed to enable a rebalancing of wealth and power between nations.

5.5. Conclusion

One may ask: ‘Very well, and now, what do we do?’. Transforming our growth-driven societies requires multiple, complementary and concurrent approaches, and absolutely everyone can play a role therein. The good news is that there are already myriad ways that people and organizations interact to provide material wellbeing, social and psychological sustenance and environmental care in our world (Gibson-Graham et al., 2015). In many places, gift and solidarity economies prevail, as well as the philosophy of community and commons, attachment to place and territory, and small environmental footprints (Feola et al., 2021). Diverse relational values with nature and conceptions of living a good life not centred on affluence exist. These alternative ways of being and relating constitute strong counterforces to the dominant regime, and rich sources of inspiration (e.g., Global Tapestry of Alternatives).

Resilience relies on diversity. What we need is a *pluriverse*, ‘a world where many worlds fit’ (Zapatista in Romero, 2023), calling to embrace plurality rather than one-fits-all solutions (Biely & Chakori, 2025). Recognizing, engaging with and learning from these other worldviews, and ‘opening up’ policy processes – what direct democracy would radically do – is key to increase the diversity of values

and ideas, as well as equity in decision-making (Leach et al., 2010). Diversity is crucial to put down our blinders.

Another good news is that we do not have to wait for decisions to (perhaps maybe) be taken by politicians. Change is unlikely to manifest without a strong, bottom-up, movement – which is already building up. This ‘capillarity’ of change, whereby change starts at the bottom, calls on each and every one of us to get involved, to the best of our ability. Some concrete ways include: lifting pressures (e.g., resist harmful projects); building and strengthening alternatives (e.g., join or start a self-managed cooperative, establish peoples assemblies e.g. in schools, work places, sport clubs, jurisdictions); actively bringing in more people (e.g., stimulate imagination about hopeful futures (see Cork et al., 2023)); cracking the dominant worldview (e.g., organize decolonial and feminist events).

And you (politicians), if you ever happen to read these lines (who knows), beyond a necessary change in the way policies are designed (see Garrett et al., 2025), a very first step would be to channel or increase investments towards these alternative initiatives to normalize them and make them available, which in turn can shift cultural norms.

Many research avenues arise from the need for transformative changes. Land system science could delve more into the implications of such alternative economic models on land systems and their telecouplings (e.g., global potential avoided land pressure, spillovers and rebound effects of cap-and-trade regulations), as well as into transition pathways, particularly for export-dependent countries (e.g., financial and social impacts of a shift from an export-based economy to food sovereignty, concrete levers to eliminate poverty traps through mechanisms based on social and solidarity economies).

And while doing all that, most importantly, let us care. Because the process shapes the outcome.

Conclusion

Rising demands for land are driving an unprecedented global land squeeze, with heightened land use competition, soil depletion, and surging land inequality, rural poverty, and food insecurity. This thesis analysed land pressures from the global demand for cocoa, one of the major forest-risk commodities, and from global sustainability land-based targets. Several major take away points can be drawn from this work.

First, the environmental impacts and land pressures of such demands are massive. In Ghana and Côte d'Ivoire, the two major cocoa producing countries, cocoa was associated with about half of the loss of tropical moist forests, amounting to over 3 million hectares of forest replaced by cocoa over 20 years – the size of Belgium. At the global scale, sustainability targets were estimated to require almost 40% of Earth's ice-free area. Together with the expected additional demands for food, fibre and timber, they would require over 14,000 million hectares in 2050, more than the global land.

Second, current sustainability interventions lack effectiveness. Traceability of forest-risk commodities and zero-deforestation commitments face intrinsic limitations, as they are confronted with leakage; non-compliant cocoa may still find its way into markets, and non-targeted crops are displaced in forests. They tend to be designed and implemented in silos, and rarely address root causes of the ecological and climate breakdown.

Third, decreasing global land demands is an essential imperative. This calls for systemic approaches to address underlying causes which are entangled in the dominant worldview and socio-economic structures. Sufficiency-oriented economies, as proposed by the degrowth movement, can bring this change forth.

On a finite planet, caring for our relationships with the land and with others is vital, and – another good news – holds great potential for joy.

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Chapters 1 and 5

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Appendices

Appendix Chapter 2

This Appendix contains the following elements:

- Supplementary text
- Supplementary tables and figures
- Supplementary references

Supplementary text

Text S1: Detailed Methodology

1. Calculating total exports per trader & imports per importing countries

First, we calculated the total volume of cocoa (in beans equivalent) exported by distinct traders and imported by distinct importing countries in 2019, using detailed data on export shipments from 2019 (SI table S1). These data include the volume and value (freight on board value, FOB) of cocoa shipments per company and country. As a quality-check, we compared these data against the available International Cocoa Organisation (ICCO) statistics on cocoa exports in the 2016-17 and 2017-18 seasons (SI figure S1). We then converted all products to kilograms of ‘cocoa bean equivalents’, using the conversion factors recommended by UTZ’s accounting system (used for mass balance calculations) to account for waste lost in processing (SI table S1).

2. Identifying the list of intermediaries (cooperatives and licensed acheteurs) from which traders source from, and their location

To trace the cocoa back to the sub-national jurisdictions where it was produced, i.e. the department, we had to establish the links between volumes bought by traders to distinct cooperatives and the location of these cooperatives within departments. The first step was to compile a dataset on these cooperatives and their geographic location. We also compiled a list of the officially-recorded acheteurs agréés.

2.1 Cooperatives

We first identified the locations of cooperatives and their connections with traders using Mighty Earth’s ‘Cocoa Accountability Map’ (<https://www.mightyearth.org/cocoa-accountability/>). This map includes information on 4451 cooperatives across the country, listing their names and acronyms, contact information, certification status, georeferenced location, size (number of members and cultivated area), and a list of companies that are known to source from that specific cooperative (SI figure S2, table S2). The

latter list comes from self-disclosures in 2019 and 2020 by major cocoa traders and processing companies of the cooperatives from which they directly sourced cocoa. 710 of the cooperatives in the Cocoa Accountability Map could be identified as providing some of the sourcing of one or more companies. Multiple companies can simultaneously source from the same cooperative. Overall, the Cocoa Accountability Map includes 1164 cooperative-company links. The Cocoa Accountability Map is, at present, the only comprehensive compilation of the sourcing from cooperatives of the industry's main actors in Côte d'Ivoire.

2.2 *Acheteurs agréés*

Besides selling their cocoa to cooperatives, farmers also frequently sell to so-called 'acheteurs', who also aggregate cocoa from multiple holdings, and then resell to trading companies. Each year, the Côte d'Ivoire's Conseil Café Cacao (CCC) publishes a list of 'acheteurs agréés', which includes the names of those acheteurs that are licensed to buy cocoa, the departments where they are allowed to buy cocoa, and the companies for which these acheteurs operate. Exporting cooperatives and trading companies were indicated as the employers in the list of licensed acheteurs (SI table S3). However, many acheteurs work unlicensed, making up the 'indirect sourcing' of trading companies.

3. *Estimating sourced volumes*

3.1 *Cooperative size*

Companies do not disclose the volume of cocoa sourced from individual cooperatives. These volumes are therefore estimated from the number of farmers per cooperative, multiplied by their cocoa production (kg of dry beans/farmer/year). For 591 cooperatives, the trading companies or certification bodies themselves reported the number of farmers per cooperative. Where a company did not report the number of farmers they purchased from for a given cooperative, but the number was reported by other companies, the sourcing from that cooperative was assigned the mean value of the sizes disclosed by these other companies. For example, if companies X, Y and Z all buy from cooperative A and company X and Y reported 200 and 500 farmers, the link between cooperative A and company Z was assigned a cooperative size of 350 farmers (the mean of 200 and 500). Finally, for the remaining 119 cooperatives which supplied a known trading company but where the number of farmers was not known, we simulated this number for each cooperative using 1000 Monte Carlo estimates (random sampling with replacement) drawn from the set of disclosed cooperative sizes (SI figure S2B). These sizes ranged between 145 and 5394 farmers per cooperative, with a median value of 591 and a mean value of 766 farmers per cooperative.

3.2 *Cocoa production per cooperative*

Next, we estimated cocoa production per cooperative. We used data from a recent study which collected production data from 441 farms across the country (Bymolt *et al* 2018). Cocoa production per farmer (kg of dry beans/farmer/year) ranges from 17 to 5800 kg, with a mean of 1225 kg and a median of 793 kg (SI figure S3).

To assign a production volume to each cooperative-company connection, we used 1000 Monte Carlo estimates based on the KIT cocoa production per farmer dataset. This simulation is based on the whole distribution of the KIT sample to better capture the variability in the observations, allowing us to derive 95% confidence intervals for our results. For example, if a cooperative included 200 farmers, we randomly sampled 200 farmer production observations from the Bymolt *et al* dataset and summed these 200 values to obtain the cooperative production. We repeated this operation 1000 times to obtain 1000 estimates of the total production of this cooperative and then averaged it, keeping the confidence interval. This calculation assumes that trading companies

purchase the entirety of the cocoa from each of their supplier farmers (rather than each farmer selling a share to multiple traders).

3.3 Volumes of sourcing traced to cooperatives & indirect sourcing per trader

To estimate the volumes that are sourced per cooperative per trader, we collated the trader-cooperative links and estimates of cooperative size and farm production, as described above. We estimate the total sourcing traced to cooperatives per trader by summing up the volumes assigned to all of their disclosed cooperatives and/or of their acheteurs. We can then estimate each trader's proportions of *indirect* sourcing by subtracting the traced volumes from the total exported volumes (obtained from 2019 shipment data).

For trading companies that did not disclose any information on the cooperatives they source from (43 companies of 73 in total), the total exported volumes are marked as 'unknown', which means that their supply chain could be any combination of indirect sourcing and sourcing traced to cooperatives. For cocoa cooperatives which export themselves (rather than selling to a trading company, which handles export), we assume that 100% of their cocoa is sourced directly from member farmers.

3.4 Volumes of sourcing traced to cooperatives per department per trader

We assigned sourcing to each department based on the known geographical location of each cooperative and licensed acheteur. In most cases, this is a single department, though some of the larger cooperatives which export directly report sourcing from several regions across the country (SI figure S4).

We sense-check the assumption that most cooperatives source the entirety of their cocoa from the department where they are registered, by comparing with the data in the list of acheteurs agréés. Here, 22 cooperatives are listed as buying from at least one acheteur. Of these, 9/22 (40%) buy from acheteurs in a single department, 3/22 buy from acheteurs in two departments, and 10/22 that work in 3 or more departments. Overall, this suggests that we may miss some wider sourcing by (large) cooperatives. Since we may over-allocate some sourcing to the department where each cooperative is located, we sense-check the volume exported per department, (to ensure that exports are not greater than production), finding a local excess in only 5 departments, representing less than 1% of the national exports.

4. Estimating production per department

As there is no official publicly available data on subnational cocoa production within Côte d'Ivoire, we developed a map of the cocoa production per department (SI figure S6). We generated this map by combining (i) a remote-sensing cocoa land use map averaged over the years 2019 to 2021 developed by ETH Zürich (Kalischek *et al* 2022), (ii) a cocoa suitability map representing the suitability to grow cocoa per pixel, from Schroth *et al.* (2016), and (iii) statistics on the national-level yearly produced cocoa volume, from the International Cocoa Organization (ICCO).

The ETH cocoa land use map was produced for Ghana and Côte d'Ivoire at 10 m resolution using 2019, 2020 and 2021 Sentinel-2 satellite images and ground truthing data with over 100,000 GPS-tracked cocoa farms and over 10,000 manually labelled background polygons (Kalischek *et al* 2022) (hereafter, we further refer to this map as the '2019 cocoa map'). A probability map was first computed by taking the average of a series of models of convolutional neural networks, indicating the degree of agreement among the models. Here, we used the binary map computed with the best threshold at 0.65 according to a held-out validation set, i.e. all values above 65 % in the probability map are classified as cocoa and all others as non-cocoa areas. This map indicates a

total cocoa area in Côte d'Ivoire of 4.3 Mha, while the FAOSTAT harvested area estimate is 4.47 Mha of cocoa (average 2017-2020).

Schroth et al. (2016) produced a map of the current suitability of West Africa to grow cocoa. Based on climatic data and information on the ecophysiology and agronomy of cocoa in West Africa, each pixel (resolution of 927.66 m) is assigned a continuous value between 0 and 1, ranging from not suitable to best suitable. This data is used to infer yield variation among Côte d'Ivoire's cocoa growing departments. Evidently, many factors other than climatic suitability impact cocoa production (e.g. certification, use of fertilisers and pesticides, etc.), but since no adequate data were available on these farm-level management practices, we were restricted to using cocoa suitability.

The ICCO publishes data on the cocoa production of the main cocoa producing countries, including Côte d'Ivoire. These data are published as production volumes per cocoa season (running from October 1st to September 30th) rather than per calendar year. For comparison with 2019 custom data, we used ICCO production data from the 2018-2019 season; 2,154,000 tonnes.

We allocated the national cocoa production to each department by weighting the cocoa area (in hectares) by its relative suitability for cocoa production.

First, we calculated the 'Relative Suitable Area' (RSA) for cocoa in each department (Eq. 1):

$$RSA_d(ha) = \sum_{n=1}^i area_n * suitability_n, \text{ (Eq. 1)}$$

where d represents all departments, n represents all pixels under cocoa, $area_n$ the area covered by each pixel, and $suitability_n$ the suitability index (0-1) of each respective pixel.

Then, we calculated the cocoa production per department by multiplying the relative suitable area of the department by the cocoa production per unit of relative suitable area (in tonnes/ha) (Eq. 2):

$$production_d \text{ (tonnes)} = RSA_d * \frac{production_{ICCO}}{\sum_{d=1}^j RSA_d}, \text{ (Eq. 2)}$$

where $production_{ICCO}$ is the total cocoa production volume of Côte d'Ivoire (from ICCO statistics).

5. Estimating the number of farms sourced from per actor

We compiled the number of mapped farms reported in CFI signatories' 2018-2020 progress reports, except ECOM who do not report on Côte d'Ivoire (SI table S4), and we estimated the total number of farms these companies are sourcing from. We used 1000 Monte Carlo estimates of the number of farms needed to reach the traded volume of each company, by summing volumes randomly drawn from the set of production data per farmer (kg/farmer/year) - one farm consisting of all farmer's cocoa plots - until it reached the volume traded by the company, keeping the number of farms it represents, and repeating this 1000 times. We then extracted the average of the number of farms and the confidence interval.

6. Estimating cocoa deforestation and linking supplies with deforestation exposure

We intersected remote sensing data on **forest cover and forest cover change, and cocoa area**, to identify cocoa-associated deforestation per department. We then linked these data to our supply chain maps to calculate the deforestation embedded in 2019 cocoa exports. This 'deforestation exposure' is attributed to specific trading companies and international buyers based on the volumes they source from each department.

6.1 Cocoa and deforestation data

To calculate the **cocoa deforestation**, we intersected the 2019 **cocoa map** (Kalischek *et al* 2022) with a map of **Tropical Moist Forest (TMF)** (Vancutsem *et al* 2021) (SI figure S5). The latter is a collection of 31 maps depicting - for each year between 1990 and 2020 - the spatial extent of undisturbed forest and their changes (deforestation, degradation and regrowth), based on 30 m resolution Landsat images.

Undisturbed TMF (UTMF) is defined by Vancutsem *et al.* (2021) as a “*closed evergreen or semi-evergreen forest without any disturbance (degradation or deforestation) observed on the Landsat valid observations up to the year of analysis*”. According to the authors, though the Landsat observation period is too short to discriminate between never-cut primary forest and secondary forest older than the observation period, by documenting all the disturbances observed over the past three decades, the remaining undisturbed TMF is getting closer to the primary forest extent. Tropical moist forest classified as Degraded (DTMF) is “*a closed evergreen or semi-evergreen forest (covered by existing or regrowing trees [but excluding tree plantations]) that has been temporarily disturbed during a period of maximum 2.5 years (900 days).*”, including “*selective logging, fires, and unusual weather events (hurricane, drought, blowdown).*”

6.2 Cocoa deforestation and degradation 2000 - 2019 used for overview of current status

Using the forest change and cocoa maps, we first created a pixel-level map (Figure 4A) providing an up-to-date overview of land-use dynamics over the last 20 years including deforestation and degradation, their association with cocoa expansion, and the locations of the remaining undisturbed and degraded TMF cover, independent of the calculations made for linking deforestation exposure to supply chains. We extracted the pixels that lost their undisturbed TFM cover between 2000 and 2019, and we distinguished those detected as degraded TMF in 2019, according to Vancutsem *et al.* (2021)’s definition (here called ‘degradation’), from those that are not detected as forest anymore in 2019 (here called ‘deforestation’). Both classes were further segregated into degraded and deforested pixels detected as cocoa (‘cocoa degradation’ and ‘cocoa deforestation’) versus non-cocoa in 2019 (‘non-cocoa degradation’ and ‘non-cocoa deforestation’). Some areas were detected as undisturbed forest by the TMF map in 2019 while being detected as cocoa by the cocoa map (0.3 Mha in total). As the TMF map is a global product while the cocoa map was produced specifically for Côte d’Ivoire and Ghana, focusing on one specific land use class, and that the latter shows good accuracy, we decided to prioritise the cocoa map over the TMF map by classifying these pixels as cocoa degradation.

6.3 Cocoa deforestation 2000 - 2015 used for cocoa deforestation exposure of 2019 traded cocoa

We calculated ‘cocoa deforestation’ as the area deforested between 2000 and 2015 that was detected as cocoa in 2019 (Figure 4B, SI Figure S5). As such, we use a 15-year allocation period (2000-2015) and 4-year lag period (2015-2019). Our deforestation estimates represent the loss of undisturbed TMF - we did not include forest that was already degraded in 2000 (1.28 Mha), because our intention was to focus on ‘primary’ forest loss and to reduce the risk of counting cocoa pixels wrongly classified as degraded forest in 2000 by the TMF map. When intersecting forest loss and cocoa, the forest change map was first resampled from 30 to 10 m resolution to match the cocoa data, using bicubic interpolation (Wang *et al* 2017). We assigned deforestation to cocoa versus other land uses proportionally to the number of cocoa pixels among the nine 10 m pixels contained in the 30 m deforestation pixels. The lag period reflects the fact that cocoa trees need between 3 to 5 years to become fully productive, so deforestation events after 2015 are not expected to have contributed to the 2019 cocoa harvest.

To clarify, there are two key differences between the cocoa deforestation 2000 - 2019 used for displaying the land use dynamics in the country, and the cocoa deforestation 2000 - 2015 used for the supply chains analyses: 1) the time period and 2) the segregation of cocoa degradation and deforestation in the first, whereas both are merged into ‘cocoa deforestation’ in the 2000-2015 map.

In both cases, as we only have a map of cocoa presence for the years 2019-2021, we are not able to tell if cocoa was the initial crop planted after deforestation, or if it came after another crop or land use (e.g., timber exploitation, mining). Similarly, forest areas classified as degraded might have experienced other land uses before cocoa establishment, such as timber logging or highly shaded food crops.

6.4 Cocoa deforestation exposure

To link cocoa sourcing to cocoa deforestation, we calculated the **relative Cocoa Deforestation** ($CD_{relative}$) per department, by weighing the annualised cocoa deforestation area per department by the 2019 cocoa production per department (which is based on the area of cocoa plantations, cf. section 4), to obtain CD values expressed in annual hectares deforested per tonne of cocoa produced (Eq. 3, figure 4B):

$$CD_{relative,d} = \frac{CD_{annual\ area,d}}{volume_{produced,d}}, \text{ (Eq.3)}$$

where $CD_{relative,d}$ is the relative cocoa deforestation for department d , $CD_{annual\ area,d}$ the area of cocoa deforestation (from 2000 to 2015) for department d , and $volume_{produced,d}$ the estimated cocoa volume produced in 2019 in department d (calculated based on the cocoa area and the suitability maps).

We calculated the **Cocoa Deforestation Exposure** linked to the sourcing traced to cooperatives of each actor (trader or importing country) ($CDE_{traced\ to\ coops}$) by multiplying the sourcing traced to cooperatives of each actor for each department (which is based on production per farmer, cf. section 3.2) by the relative CD value of the department (Eq. 4):

$$CDE_{traced\ to\ coops,actor} = \sum_{d=1}^j CD_{relative,d} * volume_{traced\ to\ coops,actor,d}, \text{ (Eq.4)}$$

where $CDE_{traced\ to\ coops,actor}$ is the cocoa deforestation exposure linked to the sourcing traced to cooperatives of a specific actor (in ha), d represents all departments, $CD_{relative,d}$ the relative cocoa deforestation of department d (in ha/tonne), and $volume_{traced\ to\ coops,actor,d}$ the cocoa volume traced to cooperatives sourced by a specific actor from department d (in tonnes).

Because actors source different volumes, we also calculated their individual annual relative CDE which provides the deforestation exposure per ton of product handled by a trader or imported by a country, per year, i.e. number of hectares deforested per tonne of sourced cocoa, annualised (divided by 15, i.e. the 15 timesteps (yearly period) of change (2000-2001, 2001-2002, etc.)). We multiplied the proportion they source from each department individually with the relative CD per department and summed these values over all departments (Eq. 5):

$$CDE_{traced\ to\ coops,relative,actor} = \sum_{d=1}^j CD_{relative,d} * proportion_{traced\ to\ coops,actor,d}, \text{ (Eq.5)}$$

where $CDE_{traced\ to\ coops,relative,actor}$ is the relative cocoa deforestation exposure linked to the sourcing traced to cooperatives of a specific actor (in ha/tonne), and

$proportion_{traced\ to\ coops,actor,d}$ the proportion (0-1) of an actor's sourcing traced to cooperatives coming from department d .

This relative cocoa deforestation exposure metric represents how prone the actors are to deforestation linked to their sourcing traced to cooperatives. It allows for direct comparison among actors as it is not scaled by total sourced cocoa volumes, but by cocoa volume unit.

Apart from their sourcing traced to cooperatives, actors are also exposed to cocoa deforestation linked to their **untraced sourcing** (i.e. indirect sourcing for companies who disclosed their suppliers, or the total unknown sourcing of companies that did not disclose anything). As it is not possible to identify the origin of untraced sourcing, the relative untraced cocoa deforestation ($CD_{untraced}$) is calculated as an average of the relative cocoa deforestation per department, weighted by the proportion of total untraced volumes supplied by each department (i.e., the remaining production in each department after accounting for sourcing traced to cooperatives) (Eq. 6). As such, the relative cocoa deforestation (ha/ton) of indirect or unknown sourcing does not vary across actors.

$$CD_{untraced,relative} = \sum_{d=1}^j CD_{relative,d} * \frac{volume_{produced,d} - volume_{traced\ to\ coops,total,d}}{volume_{untraced,total}}, \quad (Eq.6)$$

where $CD_{untraced,relative}$ is the relative untraced cocoa deforestation, $volume_{produced,d}$ the total volume of cocoa produced in department d , $volume_{traced\ to\ coops,total,d}$ the total volume traced to cooperatives and sourced from department d (i.e. over all actors), and $volume_{untraced,total}$ the total untraced volume for the whole country (i.e. over all companies and all departments).

We can use this single value of relative untraced cocoa deforestation to calculate each actor's cocoa deforestation exposure; by multiplying this value with their unknown or indirectly-sourced exported or imported volumes (Eq. 7):

$$CDE_{indirect\ or\ unknown,actor} = CD_{untraced,relative} * volume_{indirect\ or\ unknown,actor,total} \quad (Eq.7)$$

6.5 Uncertainty in remote sensing products

Our analyses focus on results using the selected maps of cocoa area and deforestation, because of their transparent methodologies, high quality in terms of accuracy and precision (SI table S5), and open-source availability. However, to show the sensitivity of our findings to the underlying data, we repeated our analyses using two other published cocoa maps (Abu *et al* 2021, Vivid Economics 2019) and two other deforestation datasets (BNETD *et al* 2017, Hansen *et al* 2013).

Text S2: Results of sensitivity test with other datasets

To test the sensitivity of our findings to the underlying data, we repeated our analyses using two other published cocoa maps (Abu *et al* 2021, Vivid Economics 2019) and two other deforestation datasets (BNETD *et al* 2017, Hansen *et al* 2013) (SI table S8, figure S11-16). The absolute value of cocoa deforestation varies from 243,000 ha (JRC-cocoa of Abu *et al.* (2021) & BNETD) to 1.59 Mha (ETH & JRC-TMF) and the percentage of deforestation due to cocoa ranges from 14 to 41%. The cocoa deforestation exposure associated with traders and importing countries thus vary in magnitude following the various datasets, but the global ranking of actors remains unchanged (SI figure S11).

Several aspects support the use of the ETH (Kalischek *et al.*, 2022) & JRC-TMF (Vancutsem *et al.*, 2021) maps. First, the cocoa map from Kalischek *et al.* (2022) showed substantially higher

accuracy than the map from Abu et al. (2021) (SI table S5). No validation data is publicly available for the Vivid Economics' cocoa map, but its cocoa area (2.8 Mha) is much lower than the cocoa area in Kalischek et al. (2022) (4.3 Mha), the latter being closer to the FAOSTAT estimate (4.7 Mha in 2019). Second, the forest area officially reported in the BNETD report, after applying a correction for mapping biases (Olofsson *et al* 2014), differs considerably from the map available for spatial analyses (from 2.9 Mha (map) to 4.7 Mha (corrected) in 2000, and from 1.2 Mha to 2.7 Mha in 2015, without considering areas that were not forest in 1986) (BNETD *et al* 2017), thus questioning the use of the BNETD map for spatial analyses. The corrected BNETD areas are closer to Vancutsem et al. than to the GFC data which are likely to commit cocoa plantations as forest (i.e. 2000 tree cover of at least 30%, as per the national forest definition).

Supplementary tables and figures

Table S1. HS codes included in the analysis, with the total Freight on Board (FOB) and tons of cocoa for the year 2019, and UTZ conversion factor to beans.

HS6	PRODUCT	FOB (millions)	Tons	UTZ Conversion factor
180100	COCOA BEANS	3573.7369	1611556.68	1.0
180310	COCOA PASTE	623.3612	218862.27	1.2
180400	COCOA BUTTER	374.4056	83433.34	1.2
180500	COCOA POWDER	55.0432	26745.29	1.2
180200	COCOA WASTE MATERIAL			0.0

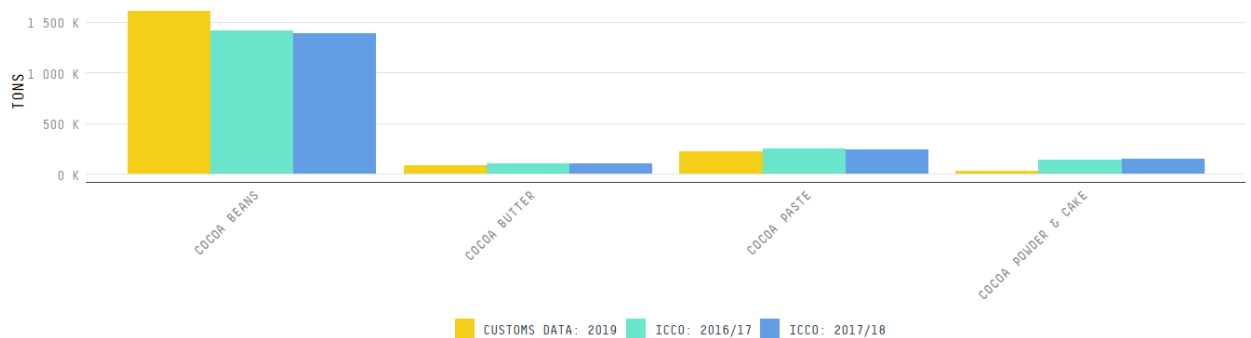


Figure S1. Comparison of 2019 shipping data and 2016/17 and 2017/18 ICCO data on exports from Côte d'Ivoire. Notes: The ICCO and the customs data match well, except that the shipping data underestimate the volumes of 'Cocoa powder and cake'. This may be because we excluded the HS code '180610' ('Cocoa powder, containing added sugar or other sweetening matter') for which the original cocoa bean content is not possible to estimate as it is mixed with other products. We also excluded 'cocoa wastes' (i.e. husks) from our analyses because values are low and to avoid double counting as cocoa waste volumes are already accounted for when using UTZ conversion factors. For the 2019 trade data, cacao waste material represented 4.79 % of the traded bean equivalent volume and 3.56 % of the FOB (i.e. the proportion of the total traded bean equivalent volume and FOB accounted for by waste products if these were included with a conversion factor of 1).

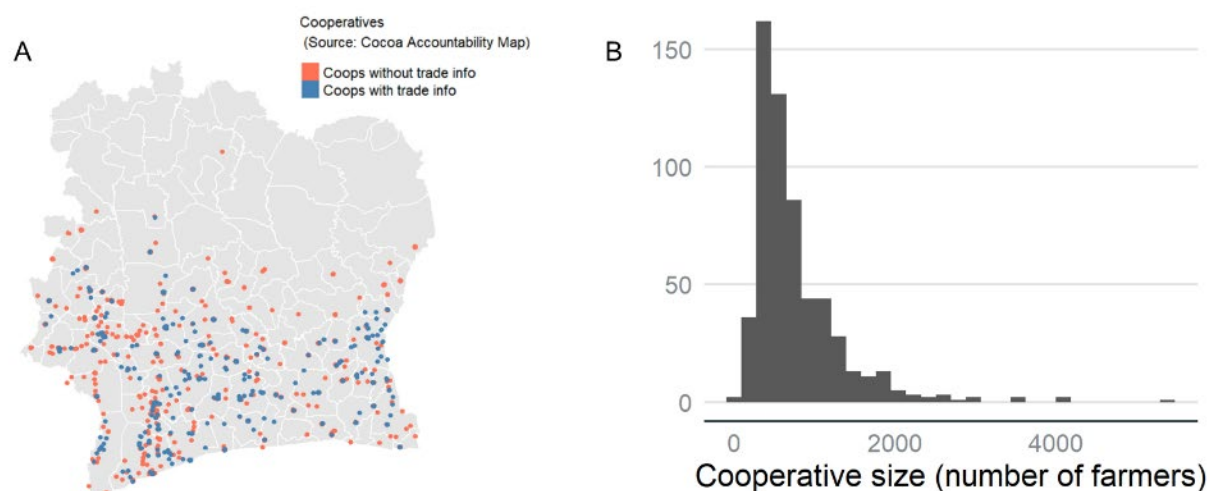


Figure S2. Cocoa cooperatives from the Cocoa Accountability Map: (A) Geographic distribution, with in blue cooperatives for which information on the companies they sell to is available, and in red those for which we do not have any trade information. (B) Distribution of cooperative sizes (number of farmers per cooperative).

Table S2. Companies included in the Cocoa Accountability Map and number of cooperatives they reported sourcing from.

Company	Number of cooperatives	Coordinates disclosed?
OLAM	198	No
BARRY CALLEBAUT	192	Yes
CARGILL	165	Yes
MARS	90	Yes
BLOMMER	79	Yes
NESTLE	76	No
FERRERO	71	No
HERSHEY	69	No
MONDELEZ	62	No
ECOM	44	No
TOUTON	34	Yes
CEMOI	33	Unknown
SUCDEN	25	Yes
COCOANECT	15	Unknown
TONY'S CHOCOLONEY	4	Yes
COCOASOURCE	3	Unknown
ALBERT HEIJN	1	Unknown
ALTER ECO	1	No
ETHIQUABLE	1	Unknown
PURATOS	1	No

Table S3. Named trading companies and cooperatives for which licensed buyers (acheteurs agréés) operate, indicated in the list of published acheteurs agréés by Côte d'Ivoire's Conseil Café Cacao (CCC).

Companies and cooperatives sourcing from Acheteurs Agréés
Biopartenaire, as a subsidiary of Barry Callebaut
CAP, Centrale d'Achat de Produits agricoles
Cyrian Côte d'Ivoire, exporting under Cyrian International
SACC, Société Agricole de Café et de Cacao, an exporting cooperative that previously bought SAF and CIPEXI

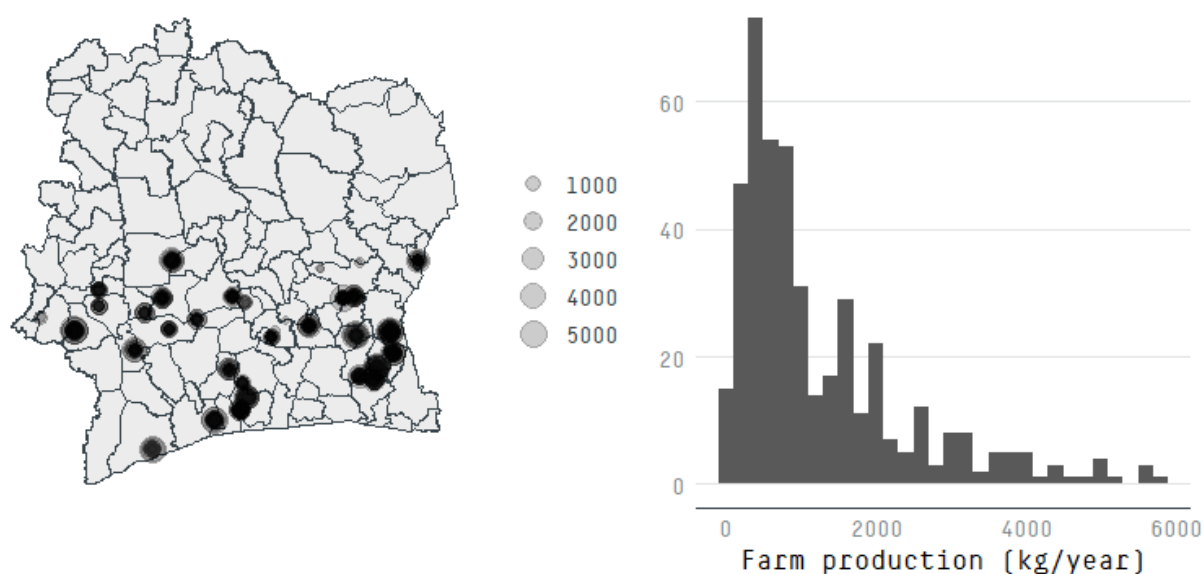


Figure S3. Distribution of observations of cocoa farmer production in kg per farmer per year from the Royal Tropical Institute (KIT) study (1 farm = 1 farmer) (Bymolt *et al* 2018).

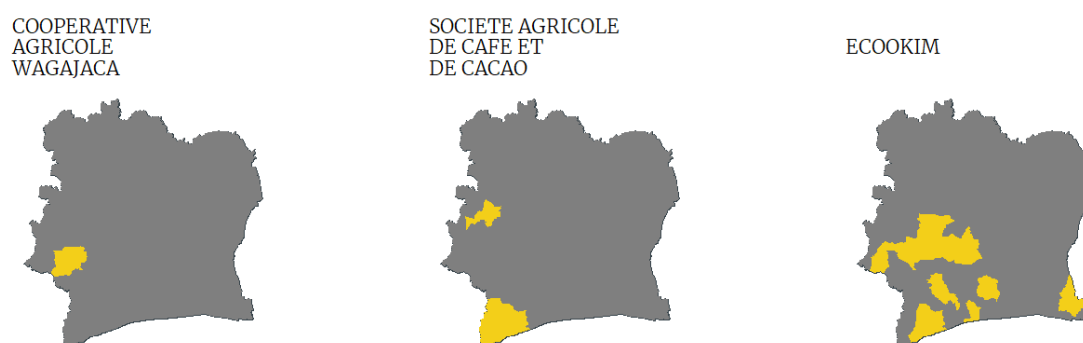


Figure S4. Examples of three larger directly exporting cooperatives sourcing from several regions across the country (COOPERATIVE AGRICOLE WAGAJACA could be linked to 2 departments, SOCIETE AGRICOLE DE CAFE ET DE CACAO (SAC) could be linked to 3 departments and ECOOKIM, which is a union of cooperatives, could be linked to 14 departments).

Table S4. Reported number of farms mapped in the 'direct' supply chain of CFI traders in Côte d'Ivoire (i.e. in the share of their sourcing coming 'directly' from cooperatives and licensed buyers) for the years 2019/20 and 2020/21, and numbers used in our study. Source: traders' published CFI progress reports.

Supplementary tables and figures

Notes: ECOM is only a signatory to the CFI for Ghana and does not report on Côte d'Ivoire. ETG joined the CFI in April 2020. Olam reported two different numbers of mapped farms in their table (71,850) and in the text (72,463) in 2020 - we use the text number here. Touton in 2020 only gave the total number of plots mapped since 2018 (not farms) - we used the ratio of the number of farms they mapped (21365) over the number of plots (25816) to get the total number of farms. The 'Total CFI' refers to the overall figures reported in Côte d'Ivoire's CFI progress reports, which include all signatories.

2019-2020 PROGRESS							Number of mapped farms in 2020 since 2018 <i>Used in study</i>	2020-2021 PROGRESS			
Number of mapped farms			% of directly sourced cocoa traceable to farm			Number of mapped farms			% of directly sourced cocoa traceable to farm (total)		
CFI Traders	# through direct investment	# on behalf of clients	Total since 2018	% through direct investment	% on behalf of clients	Total %		# through direct investment		# on behalf of clients	Total since 2018
CARGILL	24,608	45,231	NA	71%	70%	NA	69,839	24,911	76,810	NA	75%
BARRY CALLEBAUT	14,451	60,241	74,692	56%	78%	NA	74,692	40,713	NA	NA	53%
OLAM	14,269	72,463	87,324	70%	80%	78%	87,324	20,293	63,346	NA	NA
TOUTON	1942	19,423	66,245 plots	60%	92%	NA	54,824	2,814	30,799	NA	82%
SUCDEN	No numbers, only "continued to map farmers and their plots"						17,297 (2019)	21,817	NA	NA	NA
CEMOI	10,000 (13,984 in 2018/19)	NA	NA	40%	NA	NA	23,984	12,000	NA	NA	41%
ETG	1,043	14,204	15,247	92%	97%	NA	15,247	No numbers in the report			
Total CFI	354,000			74%			343,207*	465,400			72%

* Total number of farms mapped by CFI traders (without manufacturers)

Table S5. Accuracy scores of the three cocoa maps used in the sensitivity analysis (10 m resolution each), produced by ETH Zurich (Kalischek *et al* 2022), the European Commission's Joint Research Centre (JRC) (Abu *et al* 2021) and Vivid Economics (Vivid Economics 2019).

Cocoa map	Year(s)	Cocoa area (Mha)	Precision (%)	Recall (%)	F1 (%)	Accuracy (%)
ETH	2019-2021	4.3	92.6	90.9	91.7	88.7
JRC-cocoa	2019	3.7	62.2	82.9	-	-
Vivid Economics	2019	2.8	No information			

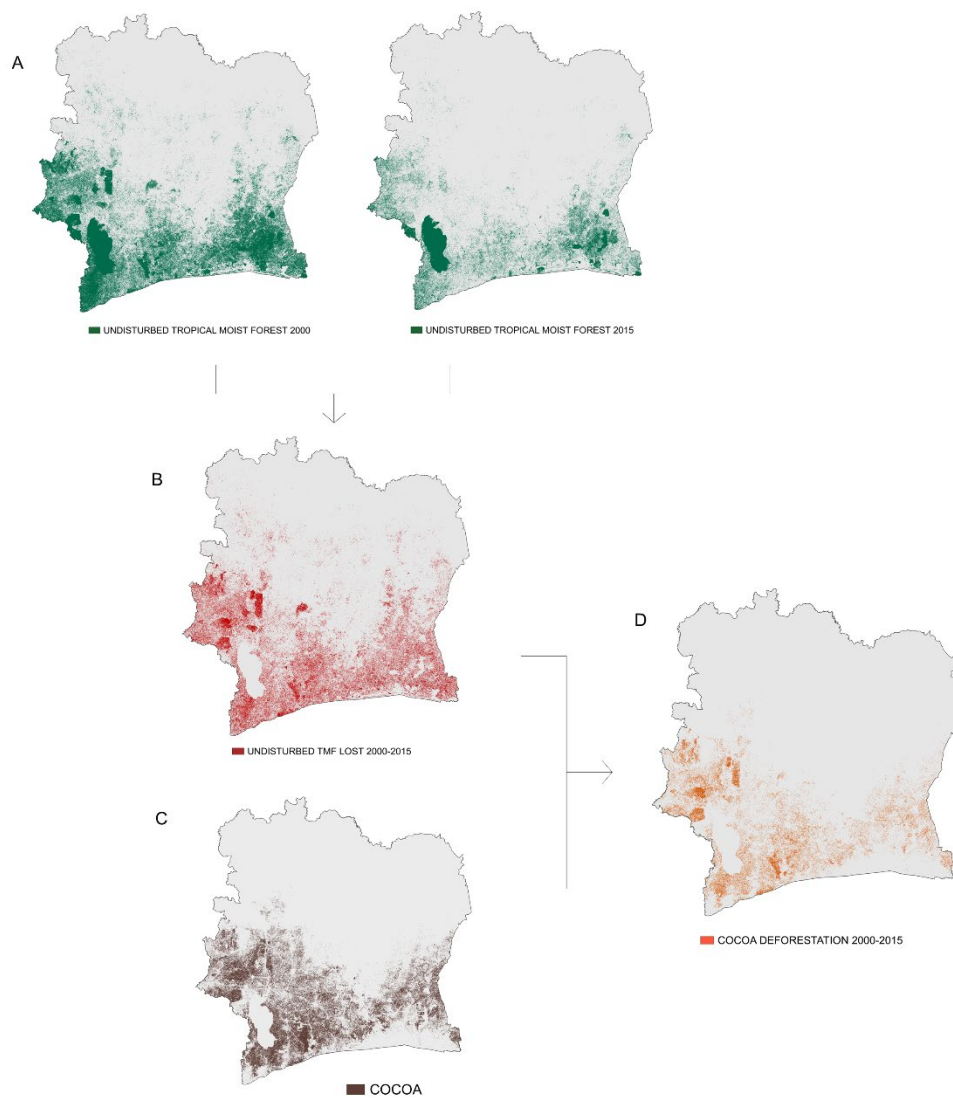


Figure S5. Methods to get the cocoa deforestation map: (A) the undisturbed Tropical Moist Forest cover maps of 2000 and 2015 (Vancutsem *et al* 2021) were combined to obtain (B) the loss of undisturbed TMF within that period, which was intersected with (C) the cocoa map (Kalischek *et al* 2022) to get (D) the cocoa deforestation over 2000 - 2015.

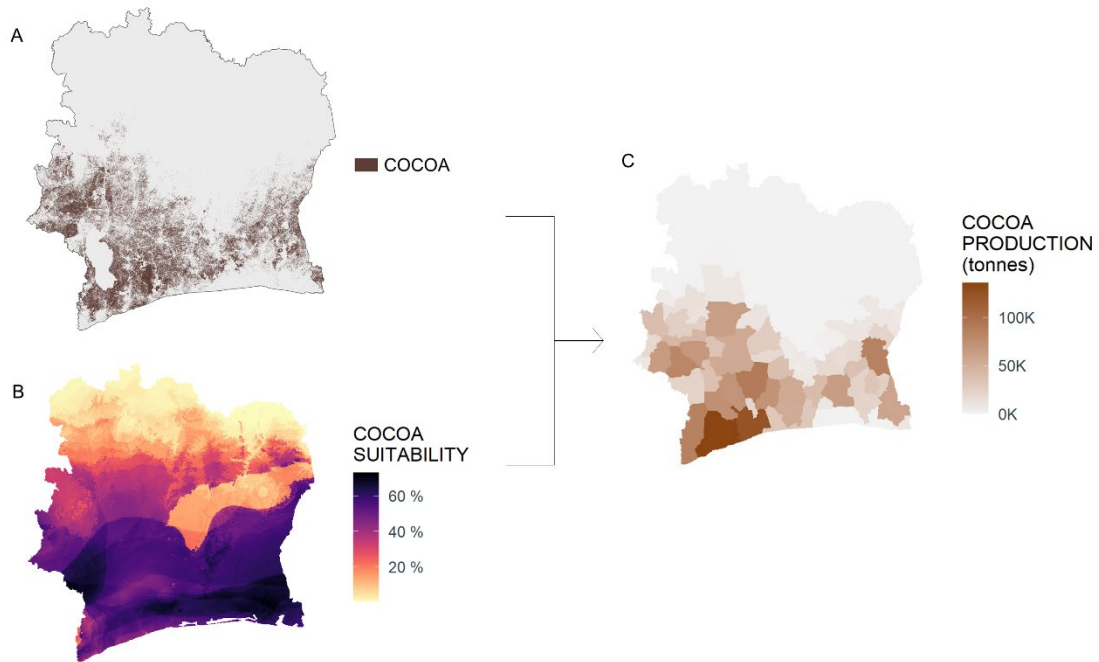


Figure S6. Methods to get the cocoa production map: (A) the 2019 cocoa map of Côte d'Ivoire (Kalischek *et al* 2022) was combined with (B) the cocoa suitability map (Schroth *et al.* 2016) to estimate (C) the cocoa production per department in 2019 by distributing the national ICCO cocoa exports following the relative suitable area per department.

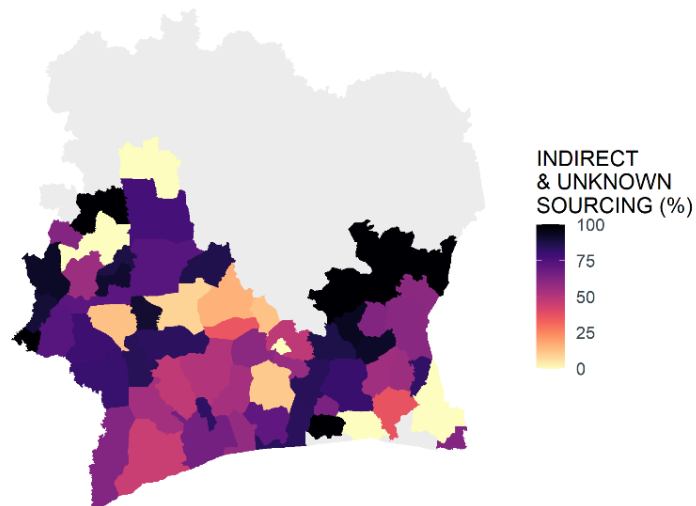


Figure S7. Percentage of untraced (i.e. indirect and unknown) sourcing in each department (only departments which produced > 500 tonnes of cocoa in 2019).

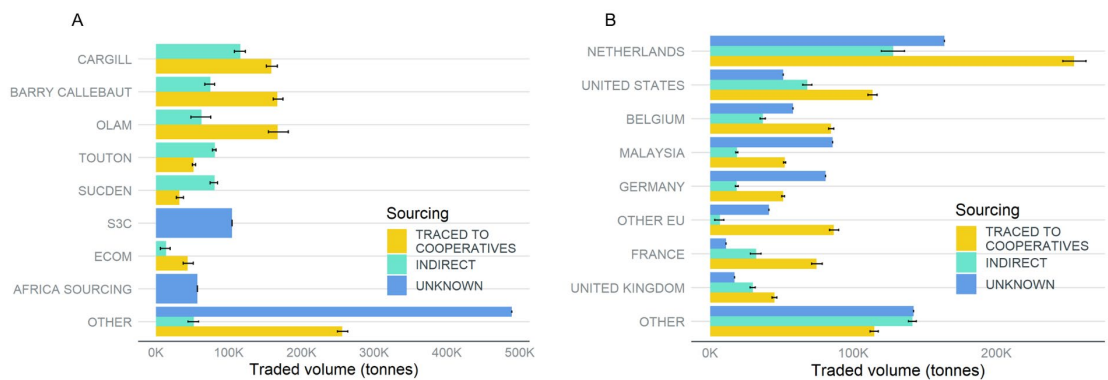


Figure S8. Share of 2019 cocoa sourcing in Côte d'Ivoire in tonnes, per sourcing type (traced to cooperatives, indirect and unknown), (A) per major trading company and (B) per importing country. Error bars represent the 95% confidence interval of the estimates.

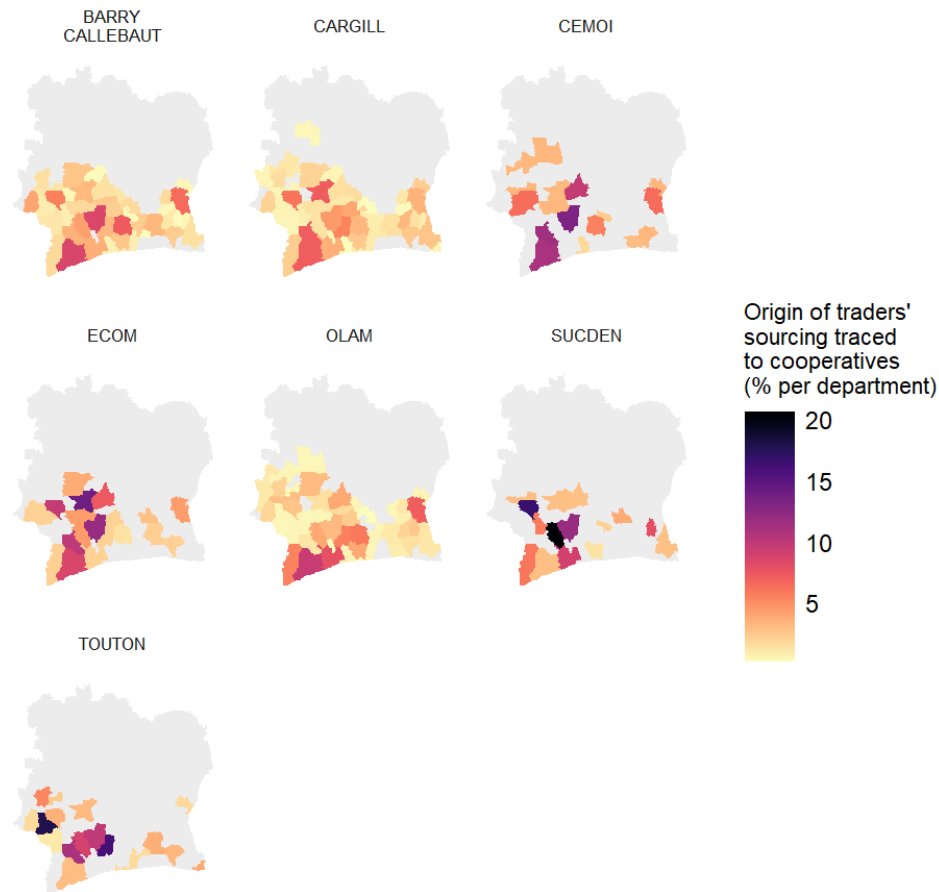


Figure S9. Percentage of cocoa sourcing traced to cooperatives per department for the major traders having disclosed their suppliers.

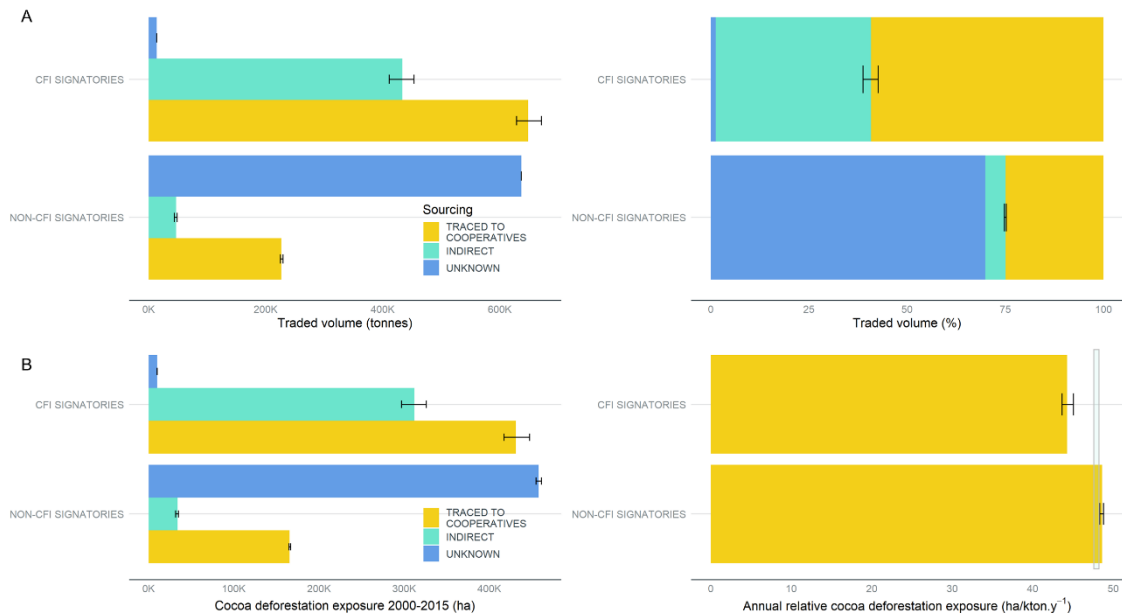


Figure S10. Comparison between CFI signatories (including ECOM, even though ECOM is only a signatory to the CFI for Ghana) and non-CFI signatories of (A) the share of the total traded volumes according to the sourcing types (i.e. traced to cooperatives, indirect and unknown) in tonnes (left) and in percentage (right), and (B) 2000-2015 cocoa deforestation exposure in hectares (left) and in hectares per kilo tonnes per year (right). Error bars represent the 95% confidence interval of the estimates.

Table S6. Deforestation and degradation data, using JRC-TMF and the ETH cocoa map, per category of protected areas (only protected areas located within the cocoa growing region of Côte d'Ivoire are considered, cf. fig.4). Source of the shapefile: Ministère des Eaux et Forêts, 2019 (Mabi-Yaya is listed as a Reserve).

Protected areas (PA) of the cocoa growing region	Total area (Kha)	Undisturbed Tropical Moist Forest (uTMF)		Degraded TMF 2019 (Kha)	Cocoa area 2019 (Kha)	Deforested & degraded 2000-2019				Deforested 2000-2019				Degraded 2000-2019			
		2000 (Kha)	2019 (Kha)			Total (Kha)	% of 2000 uTMF	Cocoa-deforested & degraded (Kha)	% of total due to cocoa	Total (Kha)	% of 2000 uTMF	Cocoa-deforested (Kha)	% of total due to cocoa	Total (Kha)	% of 2000 uTMF	Cocoa-degraded (Kha)	% of total due to cocoa
Category																	
Classified Forests	2,578	1,499	246	218	1,103	1,252	84	814	65	660	44	396	60	592	39	418	71
Parks and reserves	880	683	595	27	21	88	13	16	18	61	9	10	16	27	4	7	26
Total PAs	3,458	2,182	841	245	1,124	1,340	61	830	62	721	33	406	56	619	28	425	69
Total Côte d'Ivoire	32,302	6,710	1,523	1,467	4,358	5,187	77	2,378	46	2,803	42	1,127	40	2,384	36	1,251	52

Table S7. Comparison of the loss of forest cover between 2000 and 2015 for three forest datasets (30 m resolution each). For Côte d'Ivoire's **National Office for Technical Studies and Development (BNETD)** (BNETD *et al* 2017), forest is defined following the Ivorian forestry code of July 2014 with a minimum area of 0.1 ha, a canopy cover of at least 30% and a minimum height of 5m, and excludes agricultural

plantations. We restricted our analysis to pixels classified as ‘deforestation’ between 2000 and 2015, defined as direct human-induced conversion of forest land to non-forest land, and we did not include pixels that were not forest in 1986 (codes 121 and 122). For the **Global Forest Change dataset (GFC)** (Hansen *et al* 2013), we used the 2000 tree canopy cover dataset, defined as canopy closure for all vegetation taller than 5m in height, and we selected a tree canopy cover greater or equal to 30%, as per the Ivorian definition of forest. To get the 2015 forest cover, we used the year of gross forest cover loss (lossyear), defined as a stand-replacement disturbance or a change from a forest to non-forest state, and we kept only pixels without loss event over the 2000-2015 period. For **JRC-Tropical Moist Forest (TMF)** (Vancutsem *et al* 2021), we used the undisturbed TMF of 2000 and 2015 (cf. methods in SI text S1).

Forest maps	Forest 2000 (Mha)	Forest 2015 (Mha)	Forest loss 2000- 2015 (Mha)	% of forest loss
BNETD	2,98	1,18	1,79	60.2
GFC	14,88	13,00	1,87	12.6
JRC-TMF	6,71	3,02	3,69	55.0

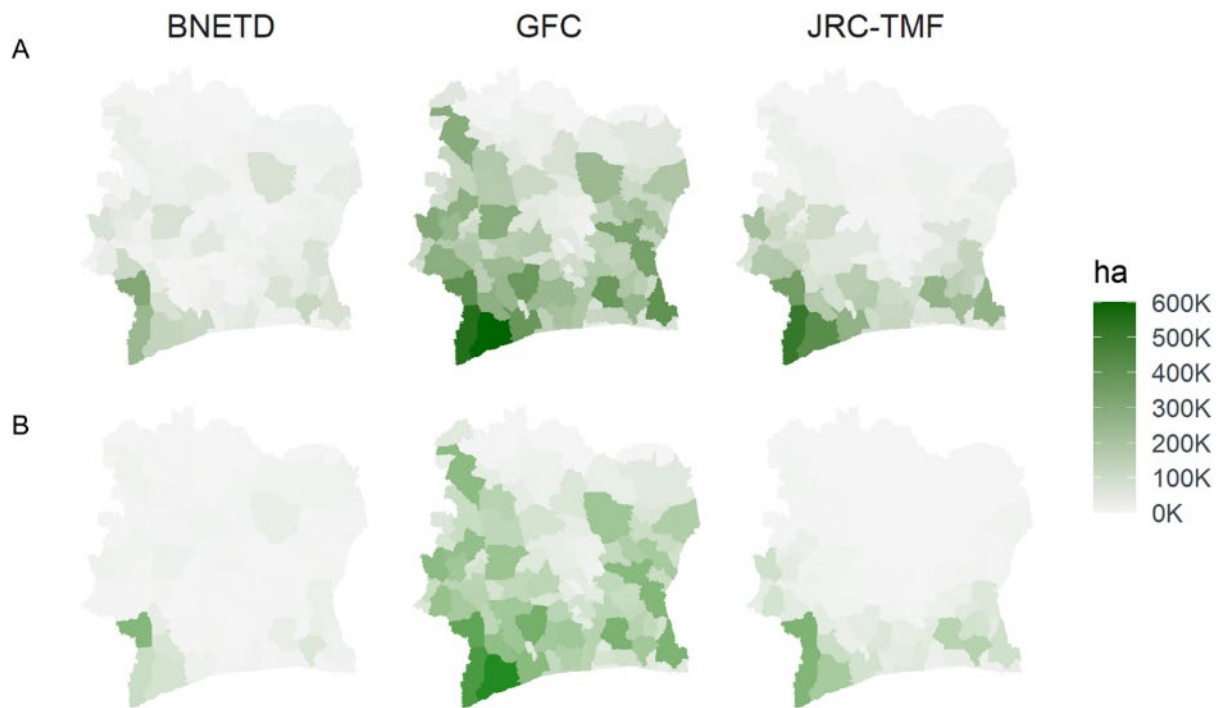


Figure S11. Forest cover area per department in hectares in 2000 (A) and 2015 (B) for the three considered forest datasets.

Table S8. Comparison of results obtained using the three cocoa maps and three forest datasets, with, for each of the nine combinations, cocoa deforestation between 2000 and 2015 in hectares, the percentage of the total deforestation due to cocoa in brackets, and the annual relative cocoa deforestation in hectares per kilo tonne of exported cocoa per year.

Cocoa maps	Forest maps		
	BNETD	GFC	JRC-TMF
ETH	710 766 ha (40%) 22 ha/kton.y ⁻¹	592 668 ha (32%) 18.3 ha/kton.y ⁻¹	1 507 443 ha (41%) 46.7 ha/kton.y ⁻¹
JRC-COCOA	370 113 ha (21%) 11.5 ha/kton.y ⁻¹	396 098 ha (21%) 12.3 ha/kton.y ⁻¹	941 173 ha (25%) 29.1 ha/kton.y ⁻¹
VIVID	430 861 ha (24%) 13.3 ha/kton.y ⁻¹	452 178 ha (24%) 14 ha/kton.y ⁻¹	994 731 ha (27%) 30.8 ha/kton.y ⁻¹

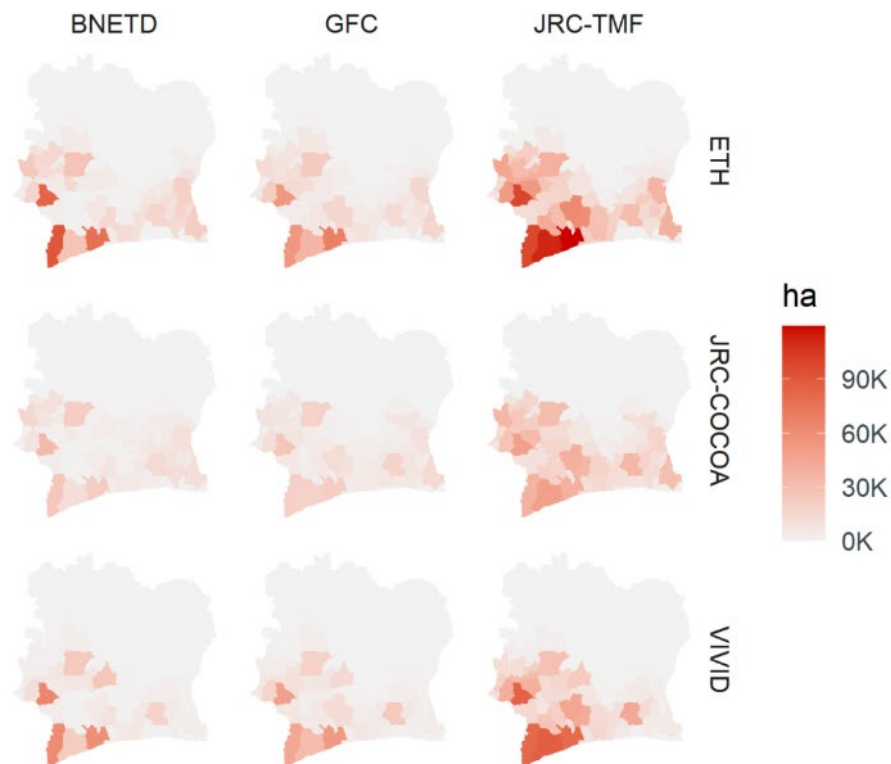


Figure S12. Cocoa deforestation between 2000 and 2015 per department in hectares for the nine combinations of the three forest datasets (BNETD, GFC and JRC-TMF) and the three cocoa maps (ETH, JRC-cocoa, Vivid Economics).

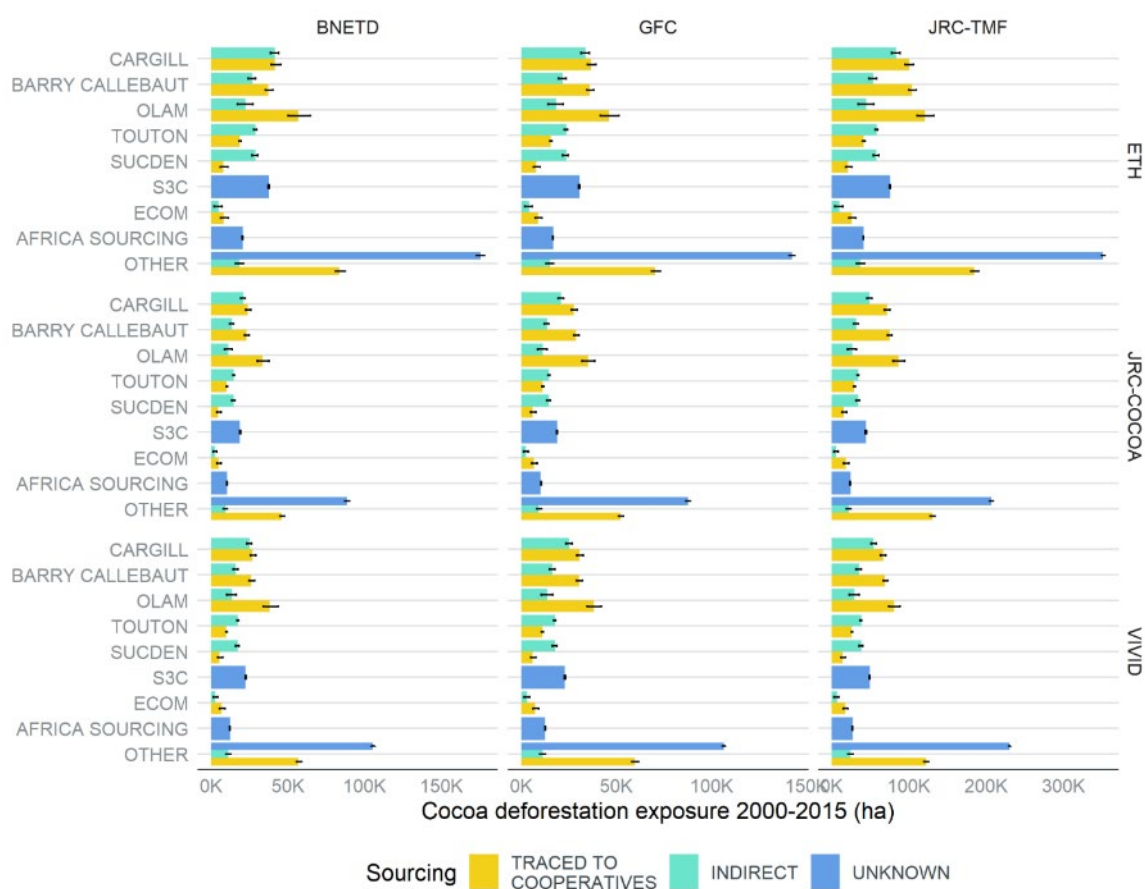


Figure S13. Sensitivity analysis: 2000-2015 cocoa deforestation exposure in hectares per trader and per sourcing type, for the nine combinations of the three forest datasets (BNETD, GFC and JRC-TMF) and the three cocoa maps (ETH, JRC-cocoa, Vivid Economics). Error bars represent the 95% confidence interval of the estimates.

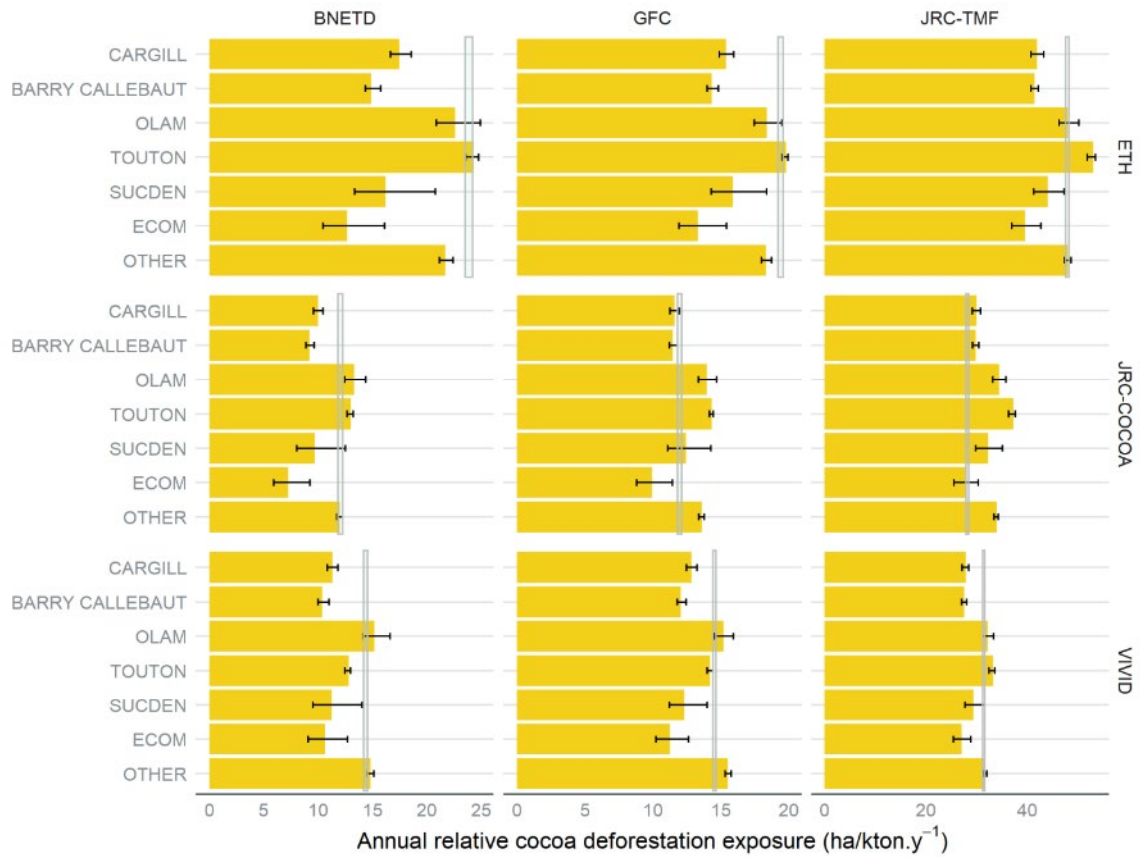


Figure S14. Sensitivity analysis: relative annual cocoa deforestation exposure (CDE) in hectares per kilo tonnes per year per trader for the nine combinations of the three forest datasets (BNETD, GFC and JRC-TMF) and the three cocoa maps (ETH, JRC-cocoa, Vivid Economics) with the vertical line representing the annual relative CDE for untraced sourcing. Error bars represent the 95% confidence interval of the estimates.

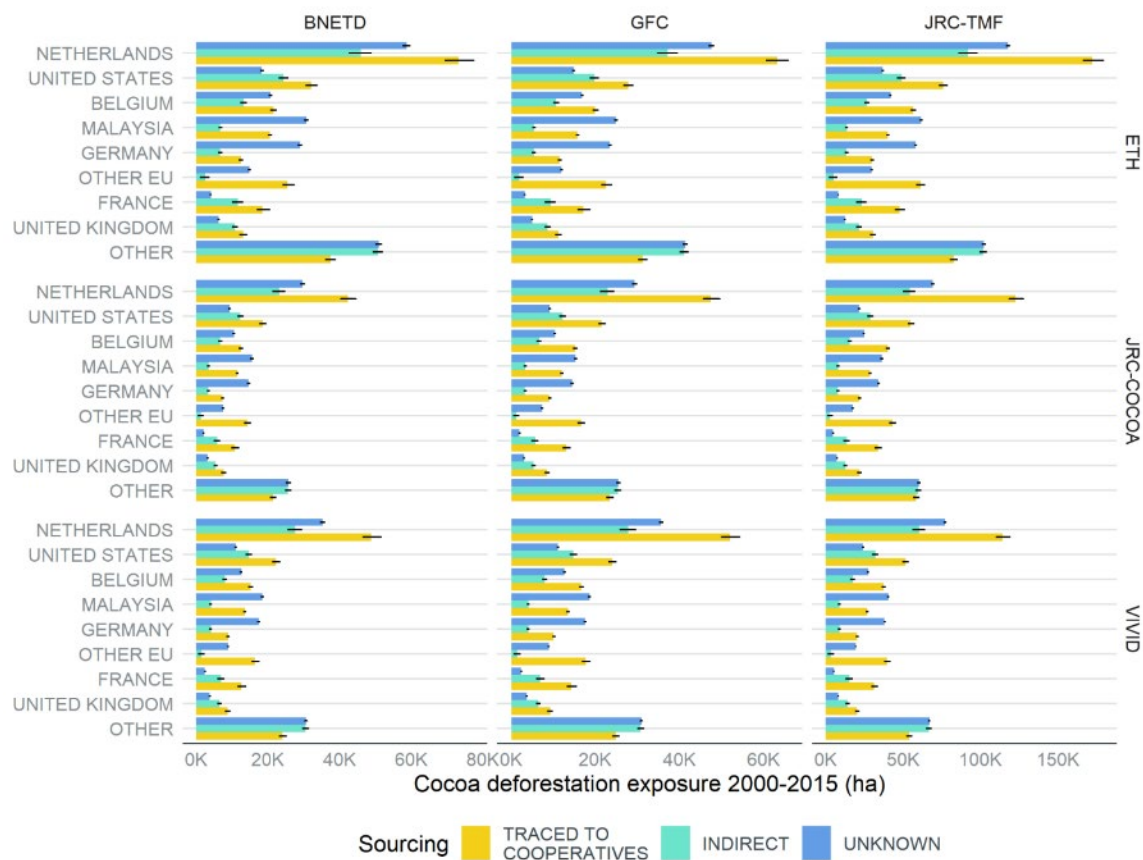


Figure S15. Sensitivity analysis: 2000-2015 cocoa deforestation exposure in hectares per importing country and per sourcing type, for the nine combinations of the three forest datasets (BNETD, GFC and JRC-TMF) and the three cocoa maps (ETH, JRC-cocoa, Vivid Economics). Error bars represent the 95% confidence interval of the estimates.

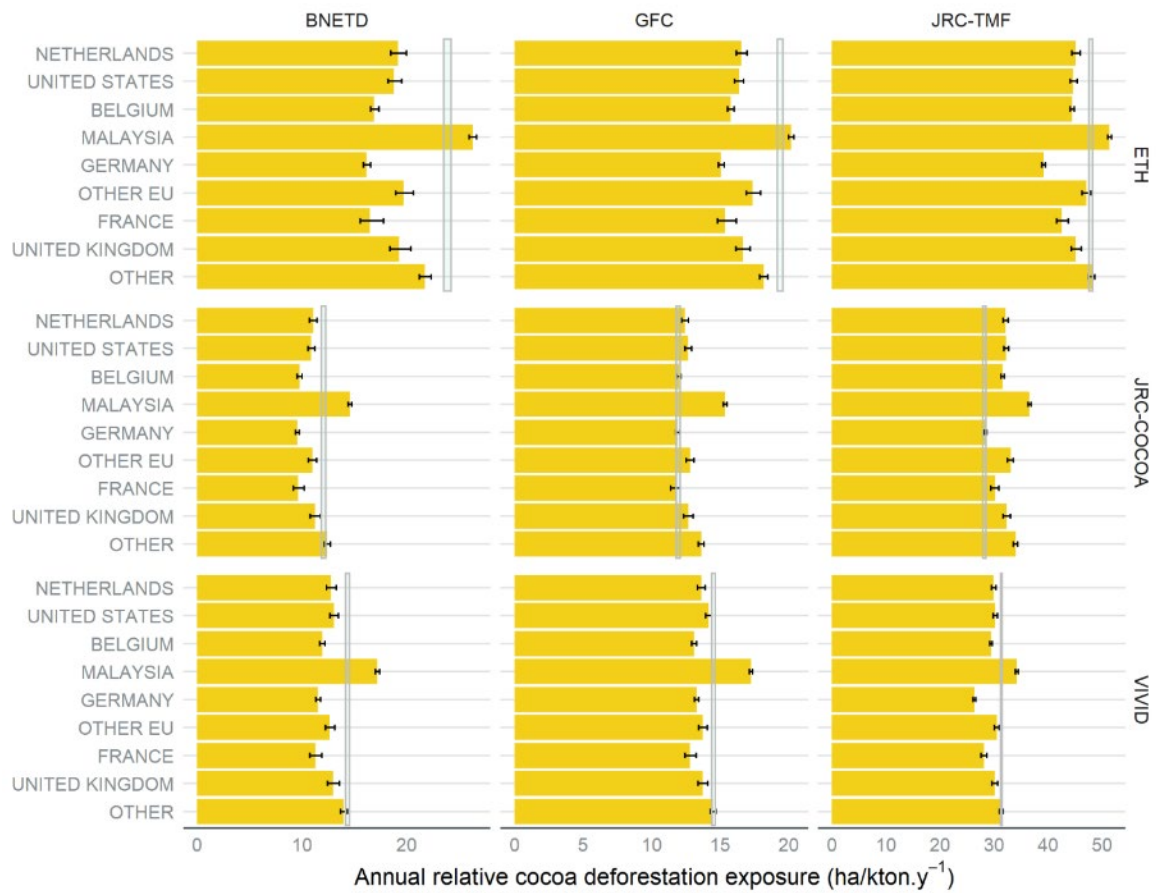


Figure S16. Sensitivity analysis: relative annual cocoa deforestation exposure (CDE) in hectares per kilo tonnes per year per importing country for the nine combinations of the three forest datasets (BNETD, GFC and JRC-TMF) and the three cocoa maps (ETH, JRC-cocoa, Vivid Economics) with the vertical line representing the annual relative CDE for untraced sourcing. Error bars represent the 95% confidence interval of the estimates.

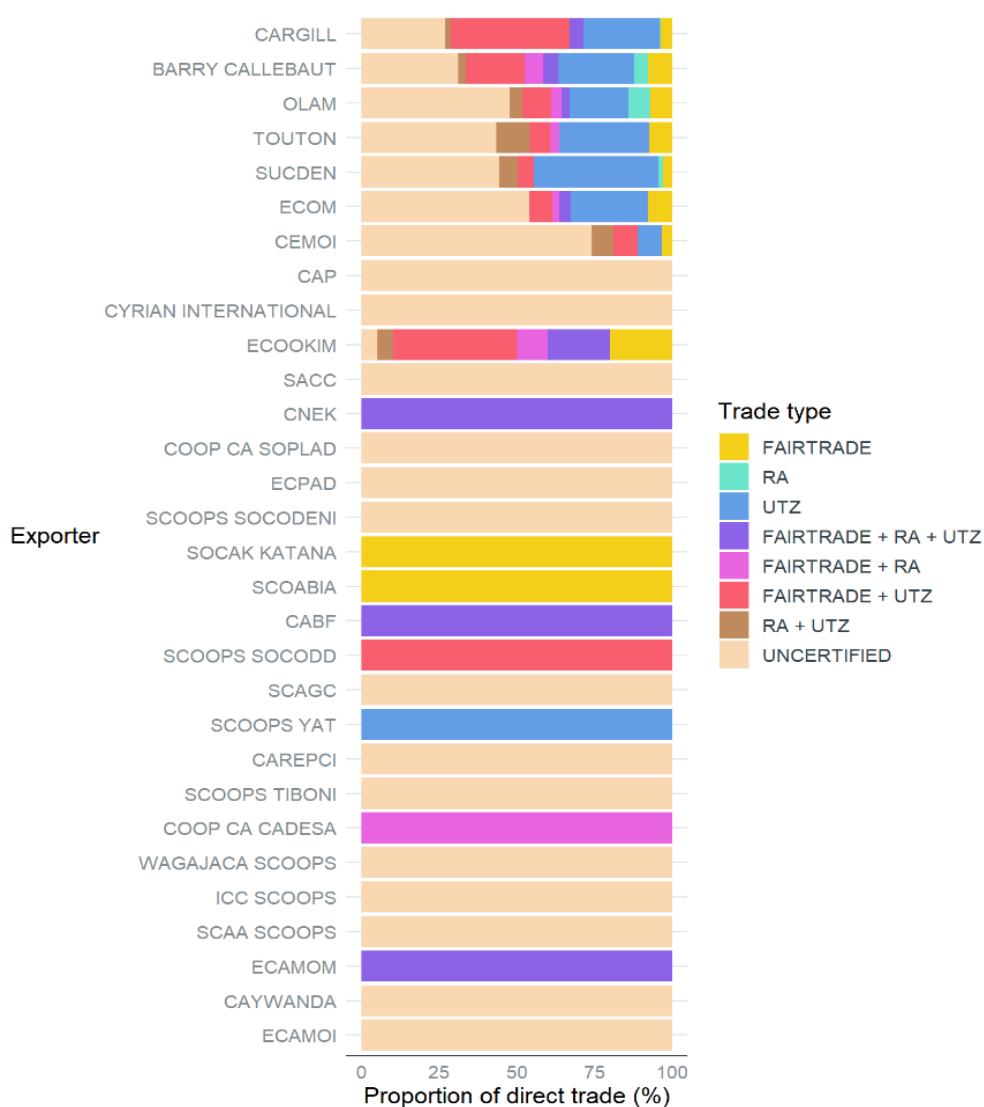


Figure S17. Proportion of 2019 traded volume under certification, per trader. (RA: Rainforest Alliance). Note: RA and UTZ have now merged. Source: disclosures by companies of the certified cooperatives they are sourcing from (from the Cocoa Accountability Map), with volumes estimated based on the methodology of our study (SI Text S1).

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Appendix Chapter 3

This Appendix contains the following elements:

- Supplementary text
- Supplementary tables
- Supplementary figures
- Supplementary references

Supplementary text

Text S1. Additional information on the methods.

S1.1. 'Direct' cocoa deforestation and degradation

We found that 383 Kha of forest was replaced by cocoa between 2014 and 2019 (45% of our estimate of direct cocoa deforestation and degradation over the 2000-2019 period). This recent deforestation and degradation (~5 years back, for cocoa to become fully productive) can be attributed with more certainty to cocoa as a direct driver (whereas earlier deforestation could have happened for other land uses, before being detected as cocoa in 2019). We note that it corresponds to 51% of the deforestation and degradation that occurred after 2014, which is similar to the overall percentage of direct cocoa deforestation and degradation that we estimate for the entire 2000-2019 period (57%).

S1.2. Maps' overlap

Mixing independent remote-sensing datasets generates errors. There is a total of 60,410 ha of overlap between the cocoa, tree plantations, mining and settlements land use maps used in our research, i.e. 1.8% of the total area of the combined land use maps in 2019 (see table S3). The overlap between cocoa and mining is the largest (~50 Kha) because the mining polygons integrate the space between mine features. (Maus et al., 2022)

Cocoa and undisturbed TMF overlapped over an area of 250 Kha in 2019 - demonstrating the difficulty to distinguish cocoa from forests. Of these, ~50 Kha (3.6% of the total forest in 2019) was detected as deforested between 2019 and 2021, suggesting that it could indeed be UTMF in 2019 which was about to be deforested and replaced by cocoa (as the cocoa map is an aggregate of satellite images for 2019, 2020 and 2021). However, it is unlikely that cocoa could already be detected only 2 years after clearing. If not detection errors, these 50 Kha could as well be the felling of shade trees in highly shaded cocoa farms (existing before 1990 which is the starting year of the JRC dataset, and which had >8 m canopy height in 2000) which might be counted as deforestation instead of a cocoa land management activity. This overlapping area was considered as already being cocoa and thus removed from the undisturbed TMF of 2000.

When overlaying maps we decided to prioritise cocoa (being a dataset developed specifically for the region, with state of the art methods and high accuracy), then tree plantations and mining (being large footprint polygons focused on a single, well-identified class), settlements (which could be misclassified mining sites), and finally TMF, as a pan-tropical map.

S1.3. Forest definitions

Undisturbed TMF (UTMF) is defined by Vancutsem et al. (2021) as a “*closed evergreen or semi-evergreen forest without any disturbance (degradation or deforestation) observed on the Landsat valid observations up to the year of analysis*”. According to the authors, though the Landsat observation period is too short to discriminate between never-cut primary forest and secondary forest older than the observation period, by documenting all the disturbances observed over the past three decades, the remaining undisturbed TMF is getting closer to the primary forest extent. Tropical moist forest classified as Degraded (DTMF) is “*a closed evergreen or semi-evergreen forest (covered by existing or regrowing trees [but excluding tree plantations]) that has been temporarily disturbed during a period of maximum 2.5 years (900 days)*”, including “*selective logging, fires, and unusual weather events (hurricane, drought, blowdown)*.”

S1.4. Cocoa yields uncertainties

To get the expansion of cocoa between 2000 and 2019 per region, we needed to estimate the area covered by cocoa plantations in 2000 at the regional level (FAOSTAT only provides the *harvested* area, at national level). As the production per region was available from COCOBOD (COCOBOD, 2023), we looked for yield estimates for 2000 in the literature to convert the production into harvested area. We picked the most comprehensive open-source dataset we could find (Ruf, 2007), to which we added an estimated unproductive cocoa area of 24% (Teal et al., 2006).

Estimating the cocoa area in 2000 faces several challenges, summarised here:

- **Cocoa can be smuggled** between Côte d'Ivoire and Ghana, thus artificially increasing or reducing production. Smuggled cocoa from Côte d'Ivoire to Ghana was estimated to explain 10% of the production increase between 2000-2005 (Ruf, 2007). However, absent robust analysis of the amount of smuggled cocoa, we were not able to account for it.
- **Collecting yields through interviews** encompasses errors. Farmers may not recall exactly the amount of cocoa produced (and they may be leaning towards under-reporting it, especially if they are under a sharecropping agreement), and the area of farmland reported may not be exact. In the Western and Central regions, farmers also use the "pole" as a unit, which has different definitions. Here, we opted for the hypothesis of one pole being equivalent to 0.36 ha, as it appears to be the most plausible estimate (Ruf, 2007, p21 and 24).
- Often, yields are calculated by dividing the farmer cocoa production by the total cocoa farmland area, without distinguishing between **productive vs. non-productive land**. We use the estimate by Teal et al. (2006, p10) of 24% unproductive land for 2001/02 (ratio between unproductive land (area of cocoa too young to produce) and land bearing cocoa (excluding trees too young to produce)). Bymolt et al. (2018, p127) also found for 2015/16 that 24% of the cocoa farmland of a farmer was 5 years old or younger on average and thus not (or barely) productive yet (63% between 5-25, 14% above 25).

S1.5. Interviews

Qualitative data were collected through two focus groups and eight semi-structured interviews, totalling 18 people covering the following groups: cocoa farmers, community leaders, NGOs, academics, government officials. SI table S6 provides details of the interviews conducted, arranged by groups of actors. The face-to-face meetings and focus groups took place during field-work in April 2022, while online interviews were conducted in 2023. Key informant for online interviews were identified based on their publications (for academics), or based on their activities (for NGOs), and were invited to participate via email. The interviews were audio recorded and detailed notes were taken. Participants were given a consent form to sign which outlined the research project, risk and benefits of participation as well as the confidentiality and anonymity of the data to encourage openness.

In person meetings with cocoa farmers and community leaders, while not specifically structured around food crop expansion into forest reserves, centred on deforestation, livelihoods, and cocoa sector policies. These discussions revealed patterns and motivations related to food crop displacement into forest reserves, providing valuable insights into the mechanisms linking cocoa expansion to food crop displacement. The online interviews were specifically designed to explore the relationship between cocoa expansion and food crop deforestation and to validate a set of hypotheses made for the analysis.

For this research, qualitative data were regarded as complementary to the satellite-based analysis and were not the focus of the work, hence bearing limitations relating to completeness. For instance, both villages where focus groups were conducted were located around the Ayum forest reserve, hence not reflecting the diversity of context and situations in the HFZ. Yet, one of the co-authors of this research, Thomas Addoah, conducted a more representative field work and conducted interviews in 77 villages, spread across eight districts in the Western North, Ashanti, and Ahafo regions, adjacent to 14 forest reserves, enabling the shaping of a more comprehensive understanding of the dynamics at play (Addoah et al., in review). Regarding the set of hypotheses (cocoa productivity in 2000, percentage of unproductive cocoa, prevalence of logging, typical land use trajectories), key informants provided consistent answers in general, though uncertainties remain, especially regarding cocoa yields and area per region in 2000.

Text S2. Additional considerations on the discussion.

S2.1. Further details on mining area and deforestation impact

- According to the last 2023 report of the Ghana Chamber of Mines, gold production (the major mined mineral in Ghana) has not increased over the past years and stayed around the 10 last years' average of 120 tonnes of gold/year. However, even with a stable production, the area affected is increasing, as one needs to move to a new area once the site is exhausted.
- CERSGIS (2025) shows an increase in area from 31 to 67 Kha between 2019 and 2024 for artisanal and small-scale gold mining only (NB: as the methods are not provided, the accuracy is unknown, and the sudden jump between 2022 and 2023 could be due to a change in detection/classification methods). As a comparison, Snapir et al. (2017) found

a mining area of 28 Kha in 2013 in the entire cocoa growing area, which they estimated to have increased to 44 Kha in 2015. Barenblitt et al. (2021) found 17 Kha of mining by 2013, and 28 Kha by 2015. They also found that while individual industrial mines leave a distinctly deep impression on the landscape, the cumulative footprint of artisanal small-scale mining was nearly 7 times larger than that of the few industrial mines in Ghana.

- This increase in area from 2019 to 2024 (+36 Kha), if added to the 140 Kha of large- and small-scale mining sites in the study area in 2019 (191 Kha in the entire country)⁴⁷ detected by Maus et al. (2022) global mining dataset used in our research, would represent an increase of about 25% in area (not accounting for the expansion of industrial sites). Depending on where this increase in land area happened (whether on forests or other land covers), the role of mining (3% of the entire deforestation and degradation over 2000-2019) would increase more or less through direct and indirect deforestation dynamics, but it is likely that the order of magnitude would remain smaller than the impact of agriculture.
- Barenblitt et al. (2021) found a total dense vegetation loss of 168 Kha from 2005 to 2019 in the southwestern part of Ghana (vs. 1.5 Mha deforested or degraded over 2000-2019 according to our results), of which 47 Kha (28%) to mining (vs. 38 Kha (3%) following our results). However, this analysis was conducted on four classes: Mining, Urban, Water and “Other”, which included *non-forested* agriculture, bare soil, and other areas of anomalous vegetation loss not attributable to the three first classes. This seems to exclude cocoa and other tree crops. Further, the land conversion was based on the variation of the spectral indicator NDVI, with the risk of mixing forests with cocoa. This results in an extremely low share of agricultural deforestation (25%).
- Giljum et al. (2022) found that the direct deforestation impact of *industrial* mining is small compared with other land-intensive activities, such as cash crops and cattle farming, in many countries with tropical forest biomes. They demonstrated that mining induces indirect deforestation in the surrounding areas of sites designated for mining activities (which we do not entirely account for in our analysis as we only look at indirect deforestation linked to food crops using a land balance approach) - though this relationship was quite low in Ghana, with 0.3% increase in deforestation when moving 10% closer to a mine. In Ghana, they found 21 Kha of deforestation driven by industrial mining between 2000 and 2019.

⁴⁷ We compared the mining area obtained from Maus et al. to the dataset published by Tang and Werner (2023) using images of ranging from 2010-2022, but the latter detects a smaller mining area of 99.8 Kha (because it limited the land cover of mining sites to the exact edges of the mine features, not including space between them) thus we did not use it for further analysis.

Supplementary tables

Table S1. Cocoa deforestation estimates from various studies for Ghana (million hectares, Mha)

Reference	Period	Coverage	Deforestation drivers and role of cocoa in deforestation	Methods
Global or continental studies providing country-level results				
Pendrill et al., 2022	2000-2018	Global / Ghana	No deforestation attributed to cocoa in Ghana. 0.02 Mha/y attributed to commodities, of which 56% to forest plantations, 32% to cattle, 10% to food crops, 1% to oil palm and <1% to rubber.	Attribute Global Forest Change (Hansen et al., 2013) tree cover loss to land use using national-level agricultural statistics
Goldman et al., 2020	2001-2018	Global / Ghana	70.6% of the total <i>agricultural</i> deforestation in the cocoa landscapes of Ghana attributed to cocoa.	Attribute GFC tree cover loss to land use using coarse land use data and yearly FAO statistics on farm area per country
Masolele et al., 2024	2000-2020	Africa / Ghana	25.4% of all forest conversion in Ghana was replaced by cocoa (4.5 Mha) and 35.6% by food crops.	Attribute GFC tree cover loss to land use using high resolution Planet-NICFI images (90% producer and user accuracy).
Singh and Persson, 2024 (pre-print)	2001-2022	Global / Ghana	500 Kha of cocoa deforestation between 2001 and 2021, representing 33% of the total 1.5 Mha of agriculture-driven deforestation they find in the entire country between 2001 and 2022.	Attribute GFC tree cover loss (modified to include only natural forests) using an hybrid model based on several remote sensing datasets and agricultural statistics
National studies				
Kyere-Boateng & Marek, 2021	NA	Ghana	The analysis identified agriculture expansion, cocoa farming expansion, illegal logging, illegal mining, population growth and policy failures and lapses as the key drivers.	Use Driver-Pressures-State-Impacts-Responses (DPSIR) analytical framework to analyse the literature-based sources of causes of deforestation and forest degradation in Ghana
Forestry Commission, 2016 (Ghana REDD+ report)	Unknown	Ghana	The principal drivers of deforestation and forest degradation are agricultural expansion (50%), wood harvesting (35%), population and development pressures (10%), mining and mineral exploitation (5%).	Unknown
European Union, 2013	1990-2008	Global study providing country-level results (p61 for Ghana)	31% of the total deforestation in Ghana (2 Mha) attributed to cocoa.	Attribute deforestation as reported in FRA 2010 reports to land use based on land use expansion (FAOSTAT)
Abu et al.,	2019	Ghana and	14.5% of the area of protected areas	Produced a Random Forest

2021		Côte d'Ivoire	are cocoa plantations in Ghana. 72% of PAs house cocoa plantations in Ghana.	model-based 2019 cocoa map (82.89% producer and 62.22% user accuracy)
Kalischek et al., 2023 [used in this study]	2000-2019	Ghana and Côte d'Ivoire	0.2 Mha cocoa is located in protected areas in Ghana (7% of the total cocoa area and >4.5% of the overall protected area in Ghana (3.7 Mha)). 26,000 ha of forest in protected areas in Ghana was replaced by cocoa, corresponding to 13.5% of the total forest lost in protected areas (193,000 ha) since 2000.	Attribute Global Forest Change (Hansen et al., 2013) tree cover loss to cocoa using a Random Forest model-based 2019-2021 cocoa map (90.9% producer and 92.6% user accuracy)
Sub-national studies				
CFI, 2023	2019-2022	Six priority cocoa landscapes in Ghana	Out of the 17,676 hectares of forest lost in CFI's six priority areas in Ghana, 13.2% was attributed to cocoa production.	Attribute deforestation to land use based on land cover maps produced for the 6 areas over 2019-2022 (no ground-truthing)
Acheampong et al., 2019	1986-2015	Ashanti Region	No specific data for cocoa. Expansion of annual crop farms and tree crops caused 78% of the forest loss.	Attribute deforestation to land use based on land cover maps produced for 1986-2015
Ajagun et al., 2022	1986-2017	Juabeso District in the Western Region of Ghana	Forests decreased from 1085 km ² to 589 km ² , with the vast majority (~90%) converted to cocoa, and croplands experienced a decline from 57 km ² to 28 km ² , also mainly converted to cocoa and displaced in forests.	Attribute deforestation to land use based on land cover maps produced for 1986-2015
Ashiagbor et al., 2022	1986-2017	Juaboso-Bia Hotspot Intervention Area (HIA) of the Ghana Cocoa Forest REDD + Program (Western Region)	1025.9 km ² (52%) of forest had been directly converted to cocoa. Admitted farms and off-reserve deforestation accounted for about 90.71% of forest conversions to cocoa, with less than 10% cocoa deforestation happening on-reserve (and thus being illegal).	Attribute deforestation to land use based on land cover maps produced for 1986-2017
Asubonteng et al., 2018	1986-2015	Akyemansa-Kwaebibrem landscape in the Eastern Region	Forest recorded a loss of 199 km ² (17%) with 24% converted to cocoa. The main cover types converted to tree-crop are food-crop land and off-reserve forest.	Attribute deforestation to land use based on land cover maps produced for 1986-2015
Benefoh et al., 2018	1986-2015	Krokorsua Hills landscape between Western and Brong Ahafo Regions	Forest margins and low-lying spots are the usual target of deforestation activities because it is the least resisted. Forest conversion to cocoa plantation made up 54.6% and 77.8% of closed forest and open forest loss.	Attribute deforestation to land use based on land cover maps produced for 1986-2015

Table S2. Datasets used

Land use	Dataset description	Coverage	Resolution	Overall accuracy	Reference
Cocoa 2019	Cocoa map <i>GEE path:</i> projects/ee-nk-cocoa/assets/cocoa_map_threshold_065	GHA & CIV, one epoch (mosaic from 2019-2021)	10m	88.7%	Kalischek et al. 2023
Cocoa 2000	COCOBOD cocoa production data <i>Download:</i> https://cocobod.gh/cocoa-purchases	GHA, annual, 1974-2019	Region		COCOBOD 2023
Forest	Joint Research Center (JRC) map of Tropical Moist Forest (TMF) v1-2022 <i>GEE path:</i> projects/JRC/TMF/v1_2022/Annual-Changes	Global, annual maps for 2000-2021	30m	94.6% for Africa	Vancutsem et al. 2021
Tree plantations (including rubber, palm, coconut)	Joint Research Center (JRC) map of Tropical Moist Forest (TMF), tree plantations class (includes includes 2019 Oil palm (10m) (Descals et al., 2021) and 2020 coconut (20m) (Descals et al., 2023) and visually delineated missing plantation areas such as rubber (cf. TMF list of updates) <i>GEE path:</i> projects/JRC/TMF/v1_2022/TransitionMap_Subtypes	Global, 2000-2021	30m	Descals coconut : 99.04 ± 0.21 % Descals oil palm: 98.52 ± 0.20 %	Vancutsem et al. 2021
Mining	Global-scale mining polygons (v2), covers the direct land used by mining activities, including open cuts, tailing dams, waste rock dumps, water ponds, and processing infrastructure. <i>Download:</i> https://doi.pangaea.de/10.1594/PAN-GAEA.942325	Global, 2019	10m	88.4%	Maus et al. 2022
Settlement 2000	World Settlement Footprint Evolution <i>GEE path:</i> projects/sat-io/open-datasets/WSF/WSF_EVO	Global, annual, 1985-2015	30m	Yet to be published	Marconcini et al. 2021
Settlement 2019	World Settlement Footprint <i>GEE path:</i> projects/sat-io/open-datasets/WSF/WSF_2019	Global, 2019	10m	~84% (preliminary results as presented in Esch et al. (2022))	Marconcini et al. 2021
Food crops	Agricultural statistics (food crop production and area by crop)	GHA, annual, 1997-2019	District		Ghana Statistical Service, upon request
Forest height 2000	GLAD Global Global Land Cover and Land Use 2000 and 2020 - Forest	Global, 2000	30m	For 2020: RMSE =	Potapov et al. 2022

	height 2000 <i>GEE path:</i> projects/glad/GLCLU2020/Forest_height_2000 from https://storage.googleapis.com/earth-enginepartners-hansen/GLCLU2000-2020/v2/download.html			6.75 m, MAE =4.76 m compared to GEDI	
Forest height 2020	ETH Global Sentinel-2 Canopy Height <i>GEE path:</i> users/nlang/ETH_Global-CanopyHeight_2020_10m_v1	Global, 2020	10m	RMSE of 7.9 m and a bias of 1.7 m compared to GEDI (16.2%)	Lang et al. 2023

Table S3. Total (unrealistic) overlaps in land uses for Ghana (ha) for each combination of remote-sensing datasets, with a total of 60,410 ha of overlap (no area has more than two overlapping layers), i.e. 1.8% of the total area of the combined land use maps in 2019.

	Cocoa	Tree plantations	Mining
Cocoa			
Tree plantations	3439		
Mining	50021	704	
Settlement	4590	126	1530

Table S4. 2000 and 2019 land use area estimates (limited to the study area) and calculation method of their 2000-2019 expansion area (Min/Mid/Max refer to the three land use expansion scenarios, with Min for the minimal expansion (consequently, with the largest 2000 area), and Max for the maximal expansion (with the smallest 2000 area))

Land use	2000 area	2019 area	Expansion area (EA)
Cocoa	Production per region averaged over 1999-2001 divided by the yields +/- 10% as max and min scenarios, +24% for unproductive area (Min: 2.08 Mha, Mid: 1.87 Mha, Max: 1.70 Mha)	Satellite-based area (2.66 Mha)	Difference between the satellite-based area in 2019 and the estimated cocoa area for 2000
Tree plantations	No data (Min: 51.1 Kha, Mid: 25.6 KMha, Max: 0 ha)	Satellite-based area (104 Kha)	Max scenario: EA = total 2019 area (i.e. considering that all tree plantations detected in 2019 were established after 2000). Min scenario: EA = 2019 area detected as forest in 2000 (i.e. considering only the area directly replacing forest over 2000-2019 as new, and thus that all the area outside forest was established before 2000).

Appendix Chapter 3

			Mid scenario: middle of the road between min and max scenarios.
Mining	No data (Min: 32.8 Kha, Mid: 16.4 KMha, Max: 0 ha)	Satellite-based area (140 Kha)	Same as tree plantations
Settlements	Satellite-based area (148 Kha)	Satellite-based area (160 Kha)	Difference between 2000 and 2019 areas
Food crops	Statistical data on food crop area per district (1.35 Mha)	Statistical data on food crop area per district (1.86 Mha)	Difference between 2000 and 2019 areas

Table S5. Cocoa yields (2001/02) used to calculate the cocoa area in 2000 (Ruf, 2007) and the number of farmers who participated in the yield estimate survey

Region	Yield (kg/ha)	-10%	+10%	# farmers
Western	356*	320.4	391.6	227
Brong Ahafo	484	435.6	532.4	52
Ashanti	181	162.9	199.1	60
Central	259*	233.1	284.9	57
Eastern	158	142.2	173.8	48
Volta	313	281.7	344.3	Average (see methods)

*with the hypothesis of one pole (i.e. unit often used by farmers in the Western Region) equivalent to 0.90 acre (1 acre = 0.4047 hectare)

Table S6. Characteristics of key informants interviewed for the research

Type	Reference	Number/Gender	Date	Location
Cocoa farmers	FG1	1 focus group discussion (2Fe-male/4Male)	April 2022	Ghana, Asunafo South District in the Brong Ahafo Region, around Ayum forest reserve
Community leaders	KI1, KI2 FG2	2 interviews (M) 1 focus group discussion (1F/3M)	April 2022	Ghana, Asunafo South District in the Brong Ahafo Region, around Ayum forest reserve
NGOs	KI3, KI4	2 M Ghana	October 2023	Online
COCOBOD	KI5	1 M	October 2023	Online
Academics	KI6, KI7, KI8	2 M Ghana 1 M international	October 2023	Online

Table S7. Tropical Moist Forest (TMF) area on- and off-reserve in the high forest zone in 2000 and 2019 (on-reserve includes here forest reserves, national parks and game reserve)

Forest type	Location	Area 2000 (Mha)	Area 2019 (Mha)	Difference 2000-2019 (Mha)
Undisturbed TMF	Total	2.6	1.1	-1.5
	On-reserve	1.2	0.9	-0.3
	Off-reserve	1.4	0.2	-1.2
Degraded TMF	Total	0.03	0.31	0.28
	On-reserve	0.01	0.15	0.13
	Off-reserve	0.02	0.17	0.15

Table S8. Deforestation and forest degradation per land use over 2000-2019 (direct and indirect), per trimmed region and aggregated at the HFZ level, with Mid/Max/Min results corresponding to the error bars of the graphs (with BA for Brong Ahafo, Def. for Deforestation and Deg. for forest degradation). Remark: Max/Min scenarios refer to the expansion of land uses, but not necessarily to the end results which depend on various factors, including the food crop area relative to the ‘other land’ area in 2000 (cf. Eq.2), with the latter depending on the upper- and lower-bar estimates of land use areas in 2000. Besides, the amount of food crop and logging deforestation and degradation is lower in Max-scenarios (vs. Min) because a higher share of the deforestation and degradation of land used for food crops in 2019 is attributed to the (larger) expansion of the other land uses (under ‘indirect deforestation and degradation’).

Region	Land use	Type	Share of deforestation and forest degradation (%)			Area of deforestation and forest degradation (ha)		
			MID	MAX	MIN	MID	MAX	MIN
HFZ	COCOA	TOTAL (DIRECT AND INDIRECT)	59.92	60.55	58.51	899,297	908,845	878,112
HFZ	COCOA	DIRECT DEFORESTATION	23.61	23.61	23.61	354,326	354,326	354,326
HFZ	COCOA	DIRECT DEGRADATION	32.88	32.88	32.88	493,515	493,515	493,515
HFZ	COCOA	INDIRECT DEF. & DEG.	3.43	4.06	2.02	51,456	61,004	30,271
HFZ	FOOD CROP	DEFORESTATION & DEG.	16.58	15.88	16.97	248,773	238,304	254,743
HFZ	LOGGING & OTHER	DEFORESTATION & DEG.	15.65	15.49	17.01	234,891	232,481	255,226
HFZ	MINING	TOTAL (DIRECT AND INDIRECT)	4.01	4.03	3.97	60,121	60,513	59,625
HFZ	MINING	DIRECT DEFORESTATION	2.54	2.54	2.54	38,108	38,108	38,108
HFZ	MINING	INDIRECT DEF. & DEG.	1.47	1.49	1.43	22,013	22,405	21,517
HFZ	SETTLEMENT	TOTAL (DIRECT AND INDIRECT)	1.02	0.97	1.09	15,276	14,433	16,305
HFZ	SETTLEMENT	DIRECT DEFORESTATION	0.13	0.13	0.13	1,881	1,881	1,881
HFZ	SETTLEMENT	INDIRECT DEF. & DEG.	0.89	0.84	0.96	13,395	12,552	14,423
HFZ	TREE PLANTATION	TOTAL (DIRECT AND INDIRECT)	2.82	3.08	2.45	42,419	46,200	36,766
HFZ	TREE PLANTATION	DIRECT DEFORESTATION	2.03	2.03	2.03	30,495	30,495	30,495
HFZ	TREE PLANTATION	INDIRECT DEF. & DEG.	0.79	1.05	0.42	11,924	15,705	6,272

Appendix Chapter 3

ASHANTI	COCOA	DIRECT DEFORESTATION	29.76	29.76	29.76	113,413	113,413	113,413
ASHANTI	COCOA	DIRECT DEGRADATION	28.91	28.91	28.91	110,178	110,178	110,178
ASHANTI	MINING	DIRECT DEFORESTATION	2.46	2.46	2.46	9,360	9,360	9,360
ASHANTI	SETTLEMENT	DIRECT DEFORESTATION	0.08	0.08	0.08	294	294	294
ASHANTI	TREE PLANTATION	DIRECT DEFORESTATION	0.05	0.05	0.05	172	172	172
ASHANTI	COCOA	INDIRECT DEF. & DEG.	0.00	0.00	0.00	0	0	0
ASHANTI	MINING	INDIRECT DEF. & DEG.	1.91	2.24	1.57	7,284	8,553	5,968
ASHANTI	TREE PLANTATION	INDIRECT DEF. & DEG.	0.07	0.09	0.05	275	354	193
ASHANTI	SETTLEMENT	INDIRECT DEF. & DEG.	1.39	1.37	1.42	5,296	5,203	5,393
ASHANTI	FOOD CROP	DEFORESTATION & DEG.	12.49	12.41	12.58	47,606	47,290	47,932
ASHANTI	LOGGING & OTHER	DEFORESTATION & DEG.	22.90	22.65	23.15	87,266	86,328	88,242
BA	COCOA	DIRECT DEFORESTATION	32.78	32.78	32.78	55,584	55,584	55,584
BA	COCOA	DIRECT DEGRADATION	13.50	13.50	13.50	22,898	22,898	22,898
BA	MINING	DIRECT DEFORESTATION	0.30	0.30	0.30	505	505	505
BA	SETTLEMENT	DIRECT DEFORESTATION	0.03	0.03	0.03	52	52	52
BA	TREE PLANTATION	DIRECT DEFORESTATION	0.01	0.01	0.01	16	16	16
BA	COCOA	INDIRECT DEF. & DEG.	4.72	5.01	4.35	8,002	8,490	7,378
BA	MINING	INDIRECT DEF. & DEG.	0.06	0.11	0.02	106	180	30
BA	TREE PLANTATION	INDIRECT DEF. & DEG.	0.01	0.02	0.00	19	33	5
BA	SETTLEMENT	INDIRECT DEF. & DEG.	0.34	0.33	0.34	569	559	581
BA	FOOD CROP	DEFORESTATION & DEG.	1.72	1.71	1.72	2,908	2,897	2,922
BA	LOGGING & OTHER	DEFORESTATION & DEG.	46.53	46.21	46.94	78,905	78,351	79,593
CENTRAL	COCOA	DIRECT DEFORESTATION	15.57	15.57	15.57	24,286	24,286	24,286
CENTRAL	COCOA	DIRECT DEGRADATION	50.42	50.42	50.42	78,625	78,625	78,625
CENTRAL	MINING	DIRECT DEFORESTATION	2.81	2.81	2.81	4,377	4,377	4,377
CENTRAL	SETTLEMENT	DIRECT DEFORESTATION	0.16	0.16	0.16	251	251	251
CENTRAL	TREE PLANTATION	DIRECT DEFORESTATION	3.07	3.07	3.07	4,782	4,782	4,782
CENTRAL	COCOA	INDIRECT DEF. & DEG.	4.59	5.45	3.30	7,156	8,502	5,148
CENTRAL	MINING	INDIRECT DEF. & DEG.	0.83	0.84	0.84	1,300	1,314	1,306
CENTRAL	TREE PLANTATION	INDIRECT DEF. & DEG.	0.68	0.82	0.52	1,054	1,272	809
CENTRAL	SETTLEMENT	INDIRECT DEF. & DEG.	1.09	1.01	1.21	1,703	1,570	1,892
CENTRAL	FOOD CROP	DEFORESTATION & DEG.	20.78	19.86	22.10	32,412	30,967	34,471
CENTRAL	LOGGING & OTHER	DEFORESTATION & DEG.	0.00	0.00	0.00	0	0	0
EASTERN	COCOA	DIRECT DEFORESTATION	18.33	18.33	18.33	37,821	37,821	37,821
EASTERN	COCOA	DIRECT DEGRADATION	34.30	34.30	34.30	70,789	70,789	70,789
EASTERN	MINING	DIRECT DEFORESTATION	1.22	1.22	1.22	2,516	2,516	2,516
EASTERN	SETTLEMENT	DIRECT DEFORESTATION	0.22	0.22	0.22	463	463	463
EASTERN	TREE PLANTATION	DIRECT DEFORESTATION	3.01	3.01	3.01	6,212	6,212	6,212
EASTERN	COCOA	INDIRECT DEF. & DEG.	0.00	0.00	0.00	0	0	0
EASTERN	MINING	INDIRECT DEF. & DEG.	0.40	0.54	0.26	826	1,107	536
EASTERN	TREE PLANTATION	INDIRECT DEF. & DEG.	1.19	1.59	0.77	2,446	3,276	1,590
EASTERN	SETTLEMENT	INDIRECT DEF. & DEG.	1.33	1.31	1.35	2,745	2,703	2,788

Supplementary tables

EASTERN	FOOD CROP	DEFORESTATION & DEG.	19.69	19.56	19.82	40,640	40,377	40,911
EASTERN	LOGGING & OTHER	DEFORESTATION & DEG.	20.31	19.92	20.72	41,926	41,122	42,758
VOLTA	COCOA	DIRECT DEFORESTATION	29.95	29.95	29.95	20,378	20,378	20,378
VOLTA	COCOA	DIRECT DEGRADATION	10.86	10.86	10.86	7,389	7,389	7,389
VOLTA	MINING	DIRECT DEFORESTATION	0.00	0.00	0.00	0	0	0
VOLTA	SETTLEMENT	DIRECT DEFORESTATION	0.07	0.07	0.07	46	46	46
VOLTA	TREE PLANTATION	DIRECT DEFORESTATION	0.03	0.03	0.03	20	20	20
VOLTA	COCOA	INDIRECT DEF. & DEG.	1.02	1.06	0.96	692	721	655
VOLTA	MINING	INDIRECT DEF. & DEG.	0.00	0.00	0.00	0	0	0
VOLTA	TREE PLANTATION	INDIRECT DEF. & DEG.	0.02	0.04	0.00	13	24	2
VOLTA	SETTLEMENT	INDIRECT DEF. & DEG.	0.14	0.14	0.14	94	94	95
VOLTA	FOOD CROP	DEFORESTATION & DEG.	18.65	18.64	18.67	12,689	12,680	12,699
VOLTA	LOGGING & OTHER	DEFORESTATION & DEG.	39.26	39.22	39.32	26,713	26,681	26,751
WESTERN	COCOA	DIRECT DEFORESTATION	19.79	19.79	19.79	102,844	102,844	102,844
WESTERN	COCOA	DIRECT DEGRADATION	39.18	39.18	39.18	203,636	203,636	203,636
WESTERN	MINING	DIRECT DEFORESTATION	4.11	4.11	4.11	21,350	21,350	21,350
WESTERN	SETTLEMENT	DIRECT DEFORESTATION	0.15	0.15	0.15	776	776	776
WESTERN	TREE PLANTATION	DIRECT DEFORESTATION	3.71	3.71	3.71	19,294	19,294	19,294
WESTERN	COCOA	INDIRECT DEF. & DEG.	6.85	8.33	3.29	35,606	43,291	17,091
WESTERN	MINING	INDIRECT DEF. & DEG.	2.40	2.16	2.63	12,496	11,251	13,676
WESTERN	TREE PLANTATION	INDIRECT DEF. & DEG.	1.56	2.07	0.71	8,117	10,746	3,673
WESTERN	SETTLEMENT	INDIRECT DEF. & DEG.	0.57	0.47	0.71	2,988	2,424	3,674
WESTERN	FOOD CROP	DEFORESTATION & DEG.	21.65	20.03	22.28	112,518	104,093	115,809
WESTERN	LOGGING & OTHER	DEFORESTATION & DEG.	0.02	0.00	3.44	80	0	17,882

Table S9. Deforested and degraded Tropical Moist Forest (TMF) area on- and off-reserve per land use in the high forest zone in 2000 and 2019 (on-reserve includes here forest reserves, national parks and game reserve)

DRIVER	OFF-RE-SERVE		ON-RESERVE		TOTAL
	ha	%	ha	%	ha
COCOA DIRECT DEFORESTATION	335,091	94.6	19,235	5.4	354,326
COCOA DIRECT DEGRADATION	474,824	96.2	18,691	3.8	493,515
MINING DIRECT DEFORESTATION	35,264	92.5	2,844	7.5	38,108
SETTLEMENT DIRECT DEFORESTATION	1,844	98.0	37	2.0	1,881
TREE PLANTATION DIRECT DEFORESTATION	23,930	78.5	6,565	21.5	30,495
REMAINING DEFORESTATION & DEGRADATION [INCL. FOOD CROP AND LOGGING AND OTHERS]	318,381	54.7	264,071	45.3	582,452

Supplementary figures

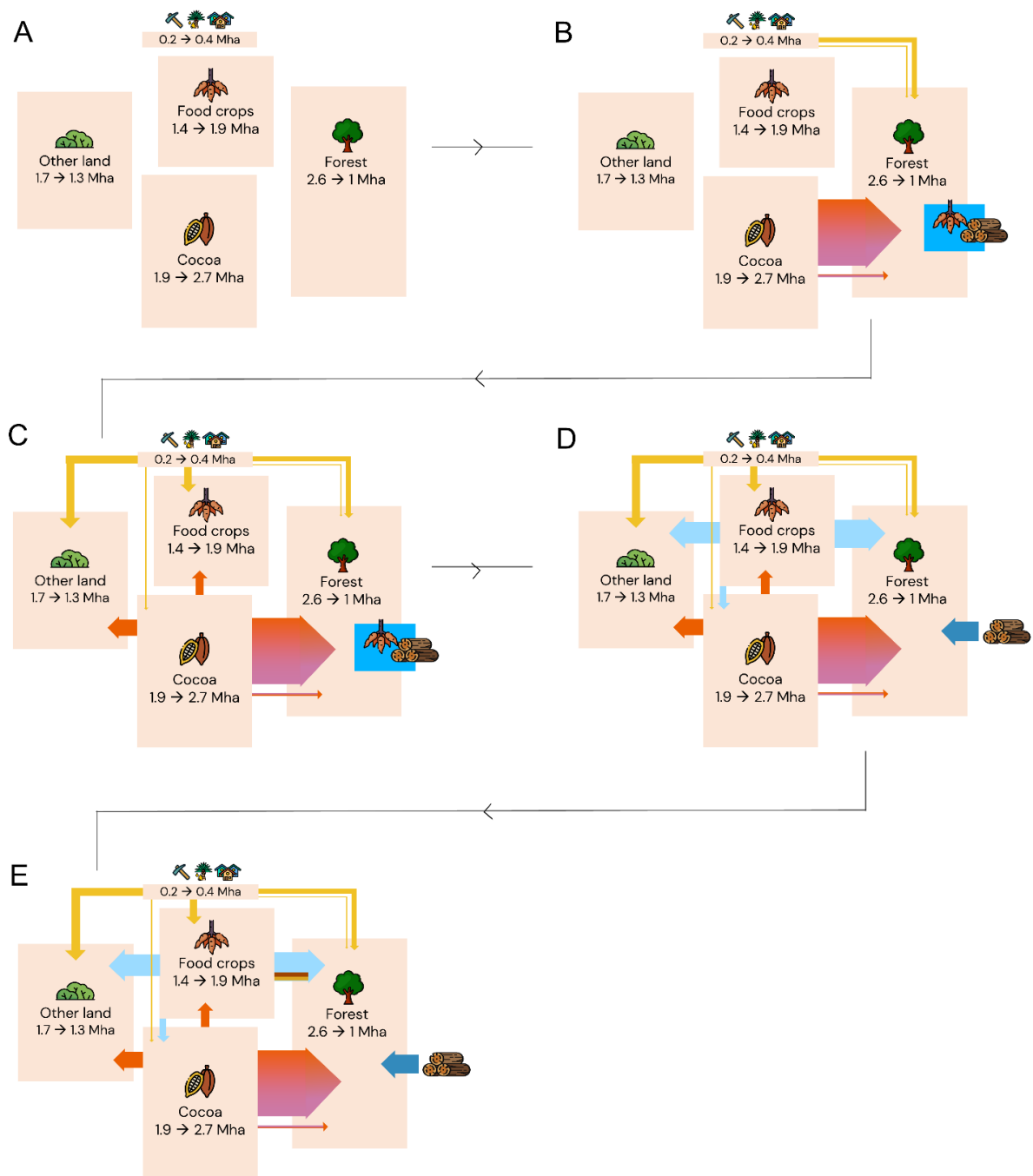


Figure S1. Schematic representation of the workflow. The width of the arrows corresponds to the area which expanded on and replaced the targeted land use (forest, food crops, other land or converted cocoa) aggregated at the high forest zone level. Box height represents the area in 2000. The numbers correspond to the area in 2000 and then in 2019.

(A) Land use areas in 2000: obtained from satellite imagery for forests, settlements, mining (mid area

represented here, i.e. mean between upper- and lower-bar estimates, cf. methods section 2.3.1), and tree plantations (mid area); calculated using 2000 yields and production for cocoa; and obtained from statistical data for food crops (cf. table S2).

(B) Expansion of each “Land Use i” (LUI) in forests over 2000-2019 (LUI is either mining, tree plantations, settlements, or cocoa): obtained by overlaying the 2019 land use maps with the 2000 forest map (thinner yellow and orange arrows correspond to expansion on protected areas – as compared to the thicker arrows for off-reserve forests) (Eq.1). All remaining deforestation and degradation (blue) as detected by the 2019 forest map can be linked to food crops, or logging and other factors (fire, fuelwood collection).

(C) Expansion of LUI outside forests over 2000-2019: the expansion outside forests (i.e. total 2019-2000 expansion minus the expansion in forests (Eq.3)) of each LUI is distributed among the available ‘pools’ of land (i.e. food crops, ‘other’ land, or converted cocoa) in proportion to their relative area within the region (Eq.2).

(D) Expansion and displacement of food crops over 2000-2019: the total net expansion of food crops (from the statistical data) plus the food crop area displaced by other LUI (yellow and orange arrows towards food crops) (Eq.4) is distributed to the various ‘pools’ of land, including forests, in proportion to their relative area within the region (once the area converted by the expansion of all LUI is subtracted) (Eq.5). The deforestation and degradation mainly due to logging is obtained by subtracting the area used for food crops in forests in 2019 (light blue arrow) to the ‘remaining’ deforestation and degradation (dark blue block in B).

(E) Indirect deforestation, i.e. share of deforestation and degradation of land used for food crops in 2019 which was displaced by LUI, represented in orange (cocoa) and yellow (mining, tree plantation and settlement combined) within the light blue arrow of deforestation and degradation of land used for food crops: calculated based on their relative expansion on food crops as compared to the total displaced and expanded area of food crops (Eq.7).

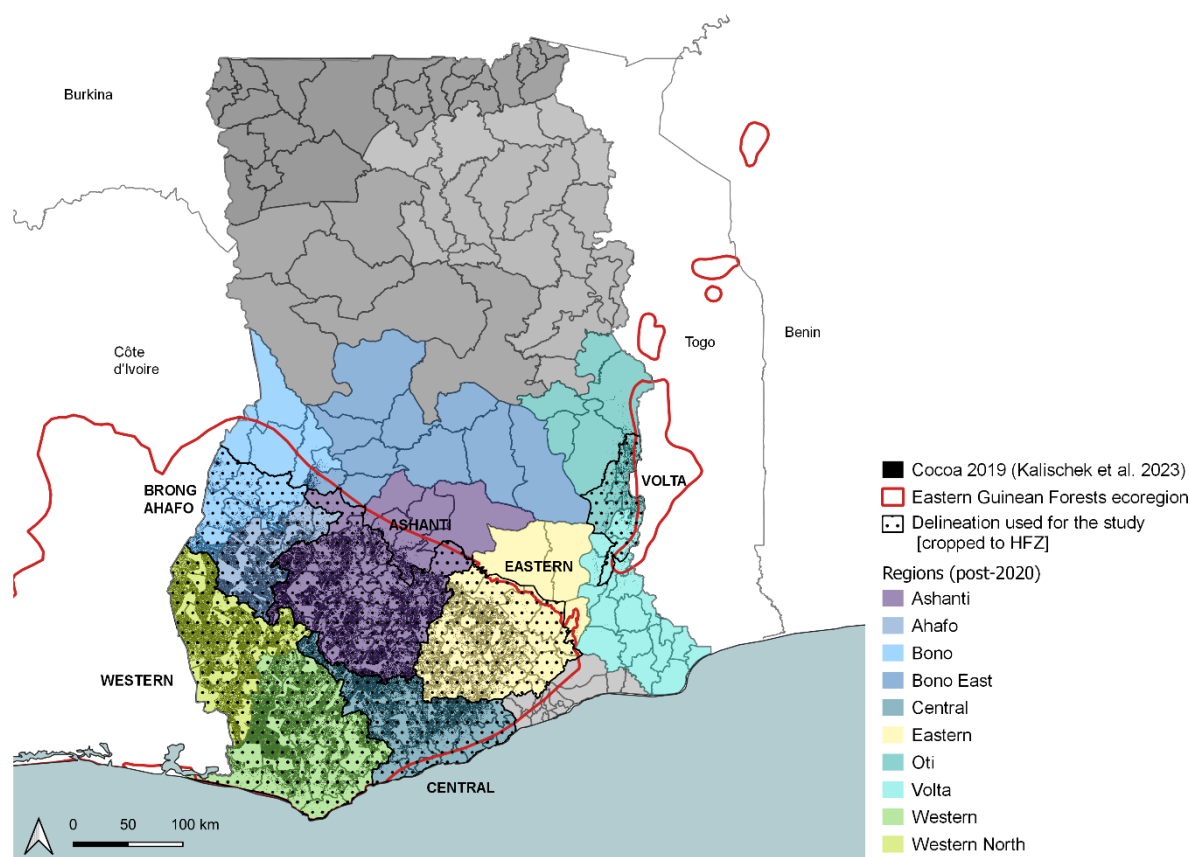


Figure S2. Map of trimmed regions used for the study (dotted). Districts located beyond the Eastern Guinean Forests ecoregions (corresponding to the High Forest Zone of Ghana) and where no cocoa was detected in 2019 were dropped. Colours correspond to the post-2020 regions, and similar colours represent the pre-2020 regions before they were split (named on the map) (blue for Brong Ahafo (including the “new” Ahafo, Bono and Bono East), green for Western (including Western and Western North), turquoise for Volta (including Oti and Volta) for which we had the cocoa production and food crop area datasets for 2000.

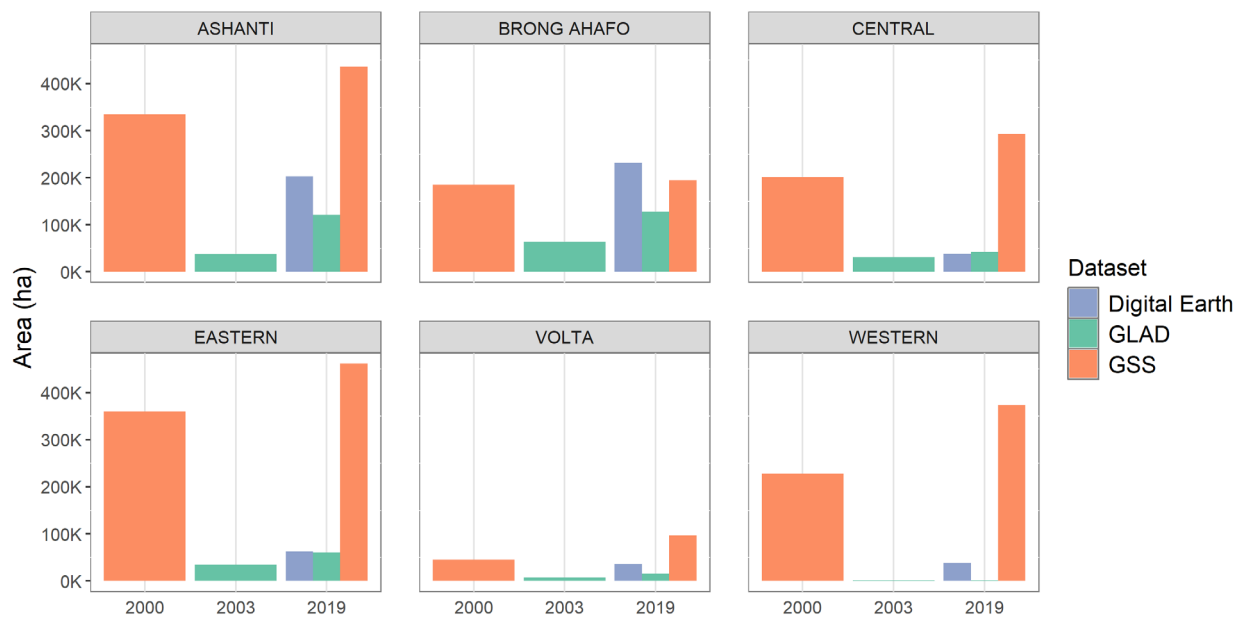


Figure S3. Comparison of food crop area in 2000, 2003 and 2019 in the high forest zone of Ghana for three datasets showing large discrepancies between the Ghana Statistical Service (GSS) dataset used in this study and satellite-based datasets (Digital Earth Africa and Potapov et al., 2022 (GLAD)). GLAD focuses on permanently cropped areas, excluding shifting cultivation areas, and is thus expected to strongly underestimate cropland area in the HFZ of Ghana.

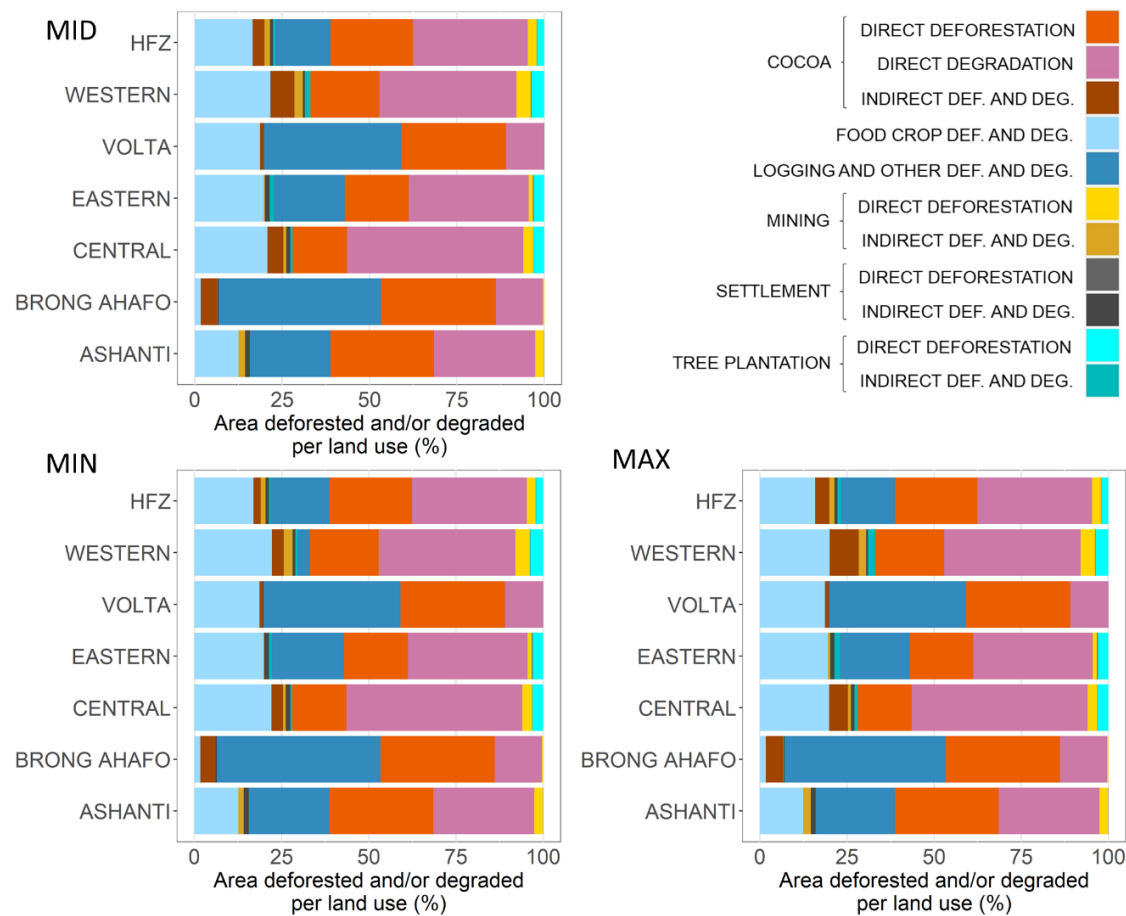


Figure S4. Land balance results for the three land use expansion scenarios, from minimum (MIN) to the maximum expansion of land uses (MAX). HFZ represents the results aggregated at the high forest zone level.

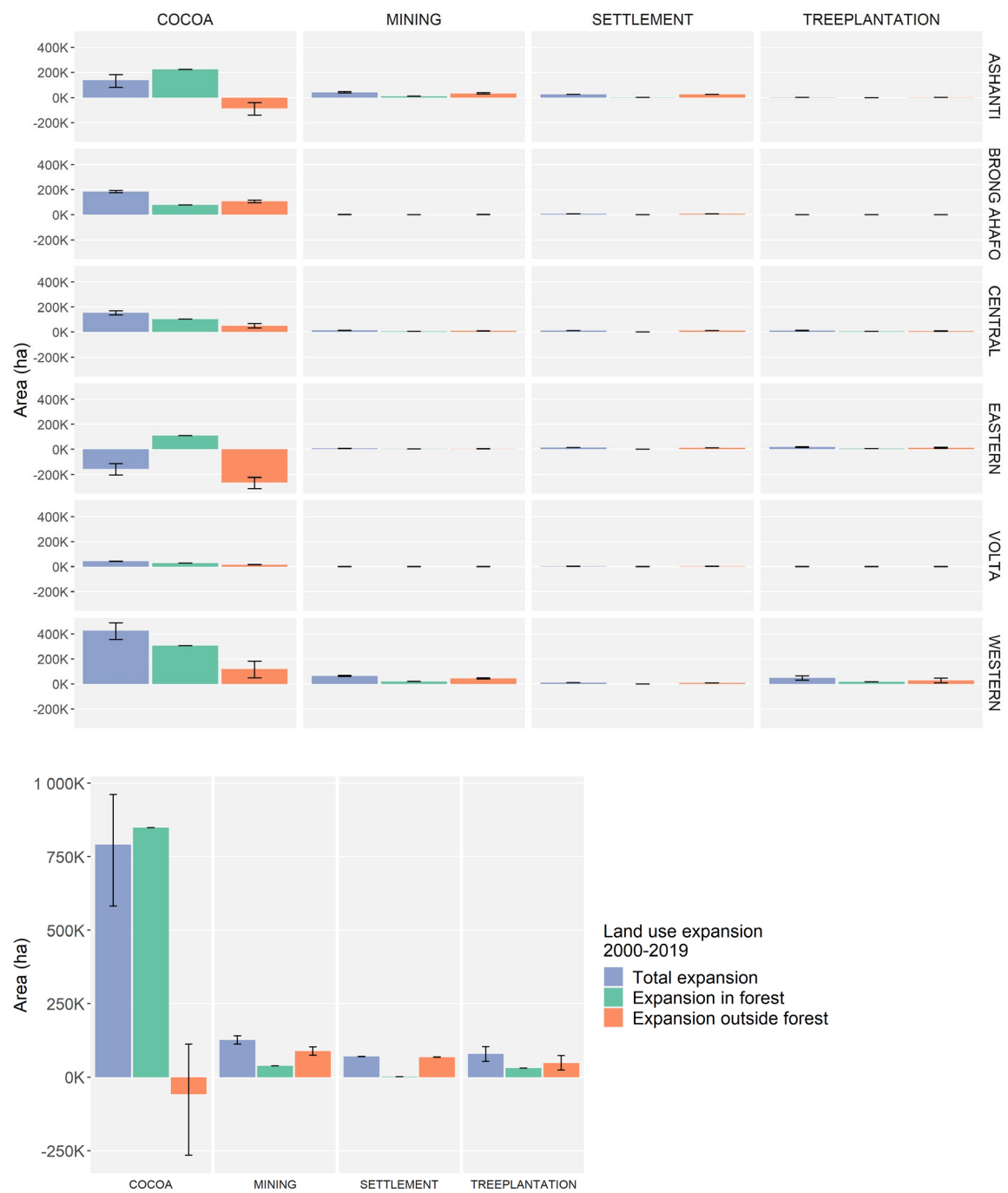


Figure S5. Land use expansion over 2000-2019, per region with the total expansion and the expansion which occurred outside and in forest (above), and at the HFZ level (below)

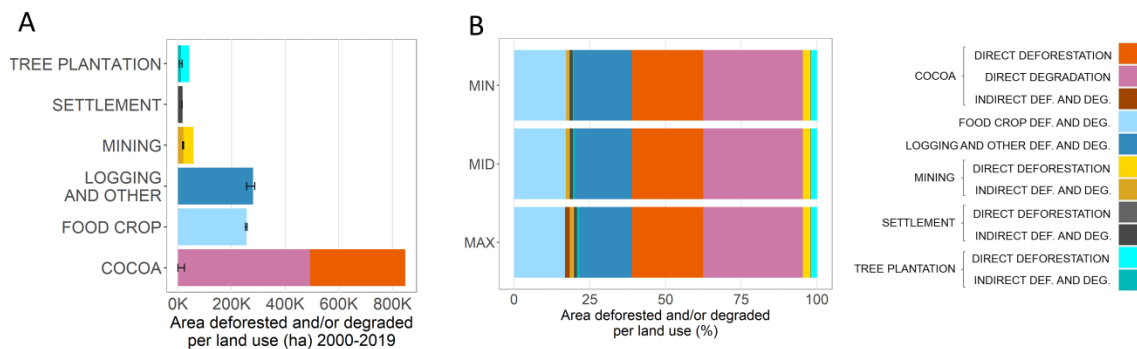


Figure S6. Results for the analysis conducted at the high forest zone level in absolute terms (A) and in percentage (B) with the minimum to maximum land use expansion scenarios (MIN-MAX), represented with the error bars in plot A for areas obtained from the land balance (i.e. LU indirect, food crop and logging and other) - as the areas obtained from satellite imagery are fixed.

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Appendix Chapter 4

This Appendix contains the following elements:

- Supplementary text
- Supplementary tables
- Supplementary figures
- Supplementary references

Supplementary text

Text S1. Material and Methods

Text S1.1. Over-selection trimming.

For each intervention, we selected the pixels per country starting from the best-suited area percentile, then going down incrementally, until the entire target area was selected.

In large countries, the last selected percentile represents a large area; if the remaining target area to cover is small, a large share of this selected percentile goes well beyond the target. Thus, we removed pixels from the last selected suitability/priority percentile per country to avoid overselection and reach the exact pledged area. We did this by first removing pixels at the border of selected areas, and if there was still overselection, random groups of 5x5 km pixels were removed within patches of selected areas.

This approach has limitations; first, dividing per percentile gives an important weight to sometimes relatively small variations, second, the last selected percentile's pixels are distributed randomly.

Text S1.2. Detailed steps of the methods.

We followed four main steps (Figure 4.5, Figure S5). The analysis was conducted using R 4.4.1 and maps for visualisation were produced using QGIS. A detailed description is provided in supplementary material (text S1).

First, we compiled quantitative targets, expressed through multilateral agreements or public commitments for each sustainability intervention, with target years either 2030 or 2050 - noting that UNCCD's LDN target was not specifically quantified or mapped as it does not clearly involve land use change or could correspond to interventions already comprised under other targets (conservation, restoration). We include countries and dependent territories in our analysis (shortened to 'countries').

(i) **Conservation:** we used CBD's GBF Target 3 of 30% land-area protected by 2030 which, absent country-specific targets, we applied to each biome within each country. By doing so, we consider that every country takes its responsibilities to conserve biodiversity in its own territory and in every biome, thus not transferring this commitment to other countries.

(ii) **Restoration**: we used governments pledges implying land use change (LUC) (not including e.g. restoration of degraded forest into healthier forest) compiled by the Land Gap report dataset (Dooley et al., 2022) - for the 102 countries without publicly-available pledge, we extrapolated the country area percentage pledged for LUC restoration by the other countries, assuming they would contribute to the CBD's Target 2 aiming at restoring at least 30% of degraded ecosystems. We did not integrate private companies' pledges to avoid double counting, assuming that governments integrate private projects in their accounting.

(iii) **Non-biomass renewables**: absent explicit targets for most countries, we used a 100% Wind, Water, and Solar (WWS) energy system scenario in 2050, detailed in installed capacity per technology for 149 countries - we only considered land-impacting infrastructure: off-roof solar photovoltaic (pv), concentrated solar power (CSP) and onshore wind turbines (hydropower expansion is limited to existing dams) (Jacobson et al., 2022). We translated the required installed energy capacity into land area using capacity density factors obtained from the literature, retaining only estimates corroborated by several sources (see table S2) and keeping low, medium and high estimates, leaving out outliers, to calculate our confidence interval (reported in table 1, figure 4.1).

(iv) **Mining**: we used a dataset of geolocated mining projects extracting or projecting to extract Energy Transition Minerals (Owen et al., 2022) - assuming that it would capture the probable future expansion of mining areas related to the development of renewable energy. We applied a 10-km buffer around project coordinates to account for the area of influence of mining sites and for the expansion of open cuts along the years (mining sites were found to cause deforestation as far as 50-km away (Giljum et al., 2022; Sonter et al., 2020)).

(v) **Bioenergy (BE)**: we used national pledges compiled by the Land Gap report and available only for five countries. We complemented them with the global target of the Net Zero 2050 roadmap of the International Energy Agency (IEA) (IEA, 2021), presented as a conservative goal, with conventional and lignocellulosic bioenergy crops representing 28 EJ in 2050 (i.e. 8% of the global energy production in this scenario), meant for hard-to-electrify sectors. As both WWS-renewables and BE sectors are not necessarily deployed coherently, we considered them individually, pushing their cursor up to the most ambitious and conservative target respectively. The WWS scenario anticipates a halving of BAU energy demand in 2050, mainly thanks to the greater efficiency of electricity compared to combustion. We thus consider this energy mix - i.e. high WWS-renewables (presented as 100% but of a relatively low global energy demand scenario) combined with IEA's moderate BE target - to be reasonable, also because our aim is to capture major prospects of land-use demands entailed by current sustainability discourses and not to model what would actually occur.

Second, we compiled global spatially-explicit suitability and priority datasets from the literature and selected the most relevant maps for each intervention (Figure 4.5). Suitability maps focus mainly on potential yield, outcome or benefit from locating the intervention in a certain place, while priority maps (here for conservation and restoration) focus more on the ranking of locations compared to one another, and also include cost of the intervention and maximizing the benefit to cost (in the case of restoration). Here, we use both types the same way. When required, we projected the input data to the equal-area Mollweide projection and resampled them at 1 x 1 km, by applying bilinear resampling method for continuous raster data and nearest-neighbor method for discrete data.

The 'current' distribution (i.e. ~2020) of the considered intervention was removed to only consider areas for expansion. We normalised these rasters by rescaling them from 0 to 1 per country as suggested by Oakleaf et al. (2019) to discern within-countries variations which are less apparent when values are globally standardized. We reclassified the country-standardized suitability/priority rasters in percentiles, per country. Then, we calculated the area per percentiles for each country for each intervention and selected the 'best-suited' percentiles to cover the target area (see text S1.1 and 1.2). We stress that the notion of 'best-suited' is limited to what these global maps consider as such, and does not capture elements that only local assessments could comprehend - thus highlighting the limitations of such global exercises and, consequently, of our results. We also calculated the mean of the non-normalised suitability/priority values of (i) all the selected pixels and (ii) the last selected percentile, per country, to get an overview of the mean, and the lowest suitability/priority score required to fulfill the pledges - providing an idea of the 'returns' of the targeted areas.

Third, to capture potential competition with current major land uses, we used a recent global land use dataset (Matej et al., 2025) for 2020 combining cropland, grazing land, forestry, built-up land and wilderness (which we renamed 'area with minimal land use') (see text S1.3).

Fourth, we overlaid the selected areas for each target and highlighted the areas with low vs. high competing interests, identifying the share of each intervention's targeted locations that overlap with that of another intervention, and with existing land uses and covers. We calculated the intensity of overlap (i.e., how many land uses overlap in the same area) and their nature (i.e., which land uses) to inform us on (i) potential conflicts or challenges in implementing these solutions, (ii) equity implications arising from the overall land demanded by these solutions as well as the concentration of these potentials in certain regions.

We also gathered recent literature estimates of other business-as-usual land demands, retaining only scientifically-robust estimates of business-as-usual situations, excluding disruptions, transformations or policy efforts, for built-up areas, pasture, cropland and forestry for timber to provide an overview of the global land demand for the coming decades and contribute to rethinking the role of land in addressing the climate and ecological breakdown.

Text S1.3. Details per intervention.

Here, we provide further methodological details for each sustainability intervention, detailing the choices we made for the pledges, the current distribution, the identification of 'priority' areas or areas with the highest 'suitability', and the way we selected the pixels:

Conservation

- **Pledges:** We used the Kunming-Montreal COP15's global target of at least 30% of terrestrial and inland water areas effectively conserved and managed by 2030. We applied this 30% country-area conservation target to each country, assuming that each country fulfils its responsibilities. We further assumed that each country would aim at conserving the variety of biomes present in their territory by conserving 30% of each biome within each country, in accordance with the CBD decision aiming for an ecologically

representative protection; covering ‘*the full range of existing ecosystems, ecological processes and regions*’.

- **Current distribution:** We used the 2024 protected areas and OECD delineation from WDPA to crop the priority map to land not already protected. When rasterizing these to 1X1 km pixels, we miss some areas, especially along rivers (instead of 23 Mkm², we rasterized 22 Mkm²).
- **Identification of targeted areas:** We identified pixels in each country for conservation in order of ‘priority’, following the map of Jung et al. (2021) of areas of global importance for conserving terrestrial biodiversity, carbon and water (10x10 km), with species distribution split by biome (thus creating separate targets for subpopulation). We assumed that countries would protect areas which would help preserve these three aspects, rather than focusing only on biodiversity or carbon. The study used a joint multicriteria spatial optimization minimizing the number of threatened species (both terrestrial vertebrates and vascular plants, with species targets set to conserve the minimum amount of species’ habitat necessary to qualify it for the conservation status ‘Least Concern’ following IUCN Red List criteria) and maximizes carbon retention (above- and below-ground biomass carbon and vulnerable soil carbon) and water quality regulation (volume of potential clean water by river basin). We locked in already existing protected areas and masked them out of the potential for expansion of conservation areas.
- **Selection:** We set the target to 30% per biome with a total of 30% land protected at country-level, based on which we calculated the additional area required once subtracting land already protected per biome inside each country, and spatialized these areas based on the priority map. By doing so, we obtain a total target of 41.8 Mkm² in 2050, i.e. ~1.8 Mkm² higher than expected, because (i) some countries already had more than 30% of their land protected, (ii) we aimed for 30% of each biome in each country to be protected, but this can lead to slight overselection (as one or several biome(s) can already be above 30%), (iii) we underestimate the current protected percentage in some areas, because we use the rasterized WDPA layer to calculate it (cf. above), (iv) Jung et al.’s map does not include large inland lakes, thus our 30% target was calculated on land only, which also underestimated the current protected percentage per biome/country. When spatialising this target, we reach a total area of 42 Mkm².

Restoration (reforestation and afforestation)

- **Pledges:** We used the Land Gap report dataset (Dooley et al., 2022) of climate pledges for 197 UN Member countries and the EU, compiled from NDCs (UNFCCC’s Nationally Determined Contributions) or long-term strategies submitted to the UN, or other national government statements (last update of October 31 2023). The land area requirements for 142 pledges were quantified, including land and forest restoration, reforestation, and for a very small number of countries, BECCS (BioEnergy Carbon Capture and Storage) - which we used for bioenergy distribution, see below. We only kept the 89 pledges implying land use change (LUC), putting aside pledges for restoration without conversion of ecosystems (e.g., restoration of degraded forest into healthier forest). Most countries are part of the Convention on Biological Diversity (CBD) convention, except the USA and Vatican - but the USA also committed to conserving at least 30 percent of US lands and

waters by 2030 (US Departments of the Interior, Agriculture, and Commerce, Council on Environmental Quality, 2021). For the 102 countries and dependent territories without a public restoration pledge under the UNFCCC, but having restoration commitments under the CBD, we used the average of the percentages of the countries' area pledged for LUC-restoration (by countries with pledges). We applied that average to countries without pledge to estimate their country area that would possibly be involved in restoration with LUC. We limited Saudi Arabia's 200 Mha of land pledged to the 40 Mha that were promised inside the country (Nagraj, 2021). The remaining 160 Mha would target land in the Middle East and, absent country-specific targets, we did not take that area into account.

- **Current distribution:** We only accounted for the projected additional area dedicated to restoration without including previous restoration activities, noting that if they involved LUC, e.g., restoration of formerly cultivated land into natural ecosystems, they are either formalized through conservation areas and thus recorded as such, or will correspond to areas of forest or grasslands in our subsequent land cover analysis.
- **Identification of targeted areas:** We identified pixels in each country for restoration in order of 'priority', following Strassburg et al.'s (2020) map of global priority areas for ecosystem restoration of converted land, optimized for biodiversity, carbon and costs (5x5 km). This map was produced based on a multicriteria approach for optimizing several outcomes of restoration (biodiversity conservation, the mitigation of climate change and costs⁴⁸) accounting for biome-specific differences in restoration outcomes, using linear programming to optimize the spatial allocation of restoration. All lands converted from natural ecosystems to croplands or pasturelands were considered. The restoration of unconverted yet degraded natural ecosystems was not included because their restoration costs and benefits are poorly quantified. As we only consider pledges for restoration implying land use change, this map of converted land matches our purpose.
- **Selection:** Some countries (mainly Libya) did not have enough suitable land for restoration according to Strassburg's map. We thus randomly selected 5x5 km land patches to reach the target, after having masked out built-up areas and all the other spatialized interventions (assuming that they would be prioritized over restoration in this case).

Non-biomass renewable technologies

- **Pledges:** Because only a few countries have explicit renewable capacity targets in their NDCs (IEA, 2024), we used Jacobson et al. (2019, 2022) dataset for 149 countries estimating the additional installed energy capacity required to reach a 100% Wind, Water,

⁴⁸ From Strassburg et al (2020): '**Benefits for biodiversity** were quantified as the reduction in potential extinction debt from habitat loss following ecosystem restoration, assessed individually for 20,319 species of mammals, amphibians and birds. [...] Restored areas within the area of habitat of each species were assumed to become habitat for that species in the long-term, thereby reducing extinction risk. [...] **Benefits for the mitigation of climate change** were measured by long-term carbon dioxide sequestration following restoration to a reference ecosystem structure in each geographical zone, considering above- and belowground biomass and soil carbon. **Opportunity costs** from agriculture were measured by the net present value of future profits that are potentially foregone by restoring cropland or pastureland back to native vegetation. To estimate **total costs**, we added restoration implementation costs by adjusting estimates from a Brazilian study according to the labour and input costs of each country.'

and Solar (WWS) energy system in 2050 as compared to their 2022 baseline. These 149 countries represent 97% of the world population and almost 100% of global fossil fuel CO₂ emissions (Jacobson et al. 2022). This study projected the demand for end-use energy to 2050 based on the business-as-usual (BAU) scenario of IEA, in the major energy sectors. They estimated a mix of WWS electricity and heat generators to meet the aggregate demand⁴⁹ in each country based on each country's resource, while at the same time limiting the land area taken up to no more than a few percent of the country area (thus maximising non-land demanding energy generators such as on-roof solar panels), and aiming for a roughly equal share of wind and solar which are complementary energy generators. No additional conventional hydropower dams are required in this scenario (rather assuming the electrification of existing dams). Here, we only considered onshore wind mills, off roof solar photovoltaic (pv) installations and Concentrated Solar Power (CSP) plants. We translated the required installed energy capacity (total of 7,114 GW for wind (+6,279 compared to 2022), 9,325 GW for solar (+8,7218) and 124 GW (+118) for CSP) into land area using capacity density factors obtained from the literature, retaining only estimates corroborated by several sources (see table S2) and keeping low, medium and high estimates, leaving out outliers, to calculate our confidence interval (reported in table 1, figure 4.1). We used the same footprint areas as Jacobson et al. for solar PV utilities; 0.0122 km²/MW for solar PV (0.0081 - 0.0389), and for CSP; 0.0294 km²/MW (0.0190 - 0.0539). For wind, we used 0.2 km²/MW (0.05 - 0.37), which is the spacing area including the area between wind turbines needed to minimize interference between them (rather than only the direct footprint). Following Jacobson et al.'s assumption, we considered that future onshore capacity increases would be met with 5 MW wind turbines (from a current average of ~2 MW), i.e. 1 turbine per km² with the medium density factor. We spatialised the medium (mid) estimate.

- **Current distribution:**

For wind, we used openstreetmap for the current (2024) locations of wind turbines (mean of 2.12 wind turbines per 1-km² grid cell with wind turbines), which we then rasterized to 1X1 km resolution, keeping the number of turbines per cell.

For solar pv, we used the global remote-sensing dataset of Kruitwagen et al. (2021) including 68,661 polygons capturing commercial-, industrial- and utility-scale PV installations (i.e. PV generating stations in excess of 10 kilowatts nameplate capacity) at the end of 2018, totalling 4824 km² (with a mean area of 7 ha per installation and median of 1.3 ha). We rasterized it to 1x1 km, missing smaller pv installations while increasing the area to 1 km² of installations which were initially smaller, reaching a total area of 4766 km².

⁴⁹ The aggregate 2050 demand in a 100% WWS scenario is reduced due to electrifying or providing direct heat. According to this study, transitioning from BAU fossil fuels, biofuels, bioenergy, and uranium to 100% WWS reduces the 2050 annual average power demand worldwide by 54%, thanks to the higher efficiency of using WWS electricity over combustion (this is reached by reducing waste heat (37%), eliminating energy needs in mining, transporting and refining fossil fuels and uranium (11%), and improving end-use energy efficiency and reducing energy use beyond those with BAU (7%)). We note that the increase in energy required to set up and run the decarbonized energy system does not seem to be included.

For CSP, following Oakleaf et al.'s methods, we used the CSP.guru dataset providing the CSP plant geolocations in the middle of 2023 (Thonig et al., 2023). We did not keep stations without coordinates, nor decommissioned stations. We applied a buffer around each geolocation with a radius calculated based on the referenced sizes of the station (mean 2.17 km², median 1.60 km²) and solar field (mean 0.46 km², median 0.37 km²). For plants lacking at least one of these two information (47 plants over 136), we applied the mean of the other plants, giving a total of 357 km² of station and field size, reduced to 309 km² once rasterized to 1x1 km (vs. 190 km² for Jacobson et al. in 2022).

- **Identification of targeted areas:** We identified pixels in each country in order of 'suitability', following the wind, solar pv and CSP 1-km spatially-explicit global land suitability maps, referred to as 'development potential indices' (DPIs), produced by Oakleaf et al. (2019). These maps were developed using spatial multi-criteria decision analysis techniques that accounted for both resource potential and development feasibility. Areas with a sector-specific resource lower than a certain threshold were excluded (solar irradiance or wind speed), as well as areas above certain slope or elevation thresholds, and areas categorized as snow, ice, or urban. Then spatially-explicit, independent criteria were produced capturing both resource availability (yields) and development feasibility (e.g., distance to major roads). For solar and wind renewable energy sectors, yields were estimated using the general equation: $PD \cdot CFI \cdot 8760$ to estimate annual yield (MWh/km²), where PD is the sector-specific power density in MW/km², CFI is the sector-specific and spatially explicit (for the *i*th cell) capacity factor derived from the corresponding resource estimate (e.g., wind speed for wind) and defined as the ratio of expected to potential power output, and 8760 is the number of hours in a year. For wind, this equation was multiplied by a spatially-explicit air density factor (AD_{*i*}), because differences in elevation have known effects on wind power production. Areas with constraints and existing production areas were removed from the mapped resource yield. Development feasibility criteria and values were standardized from 0–1. Finally, the relative importance of the spatial criteria was weighted using Analytic Hierarchy Process (AHP), and combined into DPI maps using Weighted Linear Combination (WLC).

- **Selection:**

For the three technologies, we used the current installed capacity or area in each country matching the year for which we had mapped the current distribution (2024 for wind, 2023 for CSP, 2018 for Solar PV) to adjust the projected increase in area given by Jacobson et al. (which corresponds to 2022-2050). For wind, we removed what had already been installed by 2023 (i.e. the most recent data available at the time of writing from the same source as the one used by Jacobson et al. (IRENA, 2023)) as compared to 2022 from the projected 2022-2050 additional capacity required, per country. For CSP, we removed the expansion of CSP in 2023, compared to the area in 2022 recorded in Jacobson et al., from the expansion area for 2022-2050, per country. Conversely, we added this difference to the projected increase for solar PV, to include what had been installed over 2018-2022. Jacobson et al.'s installed capacity in 2022 corresponds to 7421 km² of off-roof solar PV area, vs. 4766 km² mapped in 2018. We added this difference to the projected area for 2022-2050 per country (i.e. 2655 km², 2% of the total solar pv area in 2050).

For wind, we considered that sufficiently isolated existing turbines (1 turbine/km²) would be repowered, from 2 MW to 5 MW, thus removing the corresponding area from the projected new area (51,000 km² removed from 1,2 Mkm²). We kept the pixels with more than 1 turbine per km² as such, assuming that either they would remain standing or would be replaced by a taller turbine (for an approximately similar total capacity) which would occupy a similar space area. Importantly, the wind DPI map showed limited suitability in several regions - as a result, 47,000 km² (4%) of the projected 1,18 Mkm² could not be mapped, most notably in India,

Mining

- We used a different approach for mining, considering the lack of data on pledges for Energy Transition Minerals and Metals (ETM) extraction. According to Owen et al. (2022), *‘there is no single or standardized set of estimates on the volume of mineral resources needed to support global changes in energy technology and infrastructure’*. This is in part due to the high level of *‘uncertainty around the underlying metals consumption scenarios’* (Boer et al., 2021). Rather than relying on companies or national commitments, we thus used the publicly available dataset of Owen et al. (2022) compiling records for 5097 geolocated mining projects extracting or projecting to extract ETMs - assuming that it would capture the probable future expansion of mining areas related to the development of renewable energy. Owen et al. identified 29 mineral and metal resources needed in the energy transition, including (i) specialty commodities used in low-carbon energy technologies (e.g. graphite and lithium), and (ii) major commodities used in low-carbon energy infrastructure (aluminum, copper, iron, lead, nickel and zinc). The geolocation data was sourced from the S&P Capital IQ Pro database, one of the most comprehensive and up-to-date sources of mining data, with 36,395 geolocated mining projects worldwide at all development stages in 2021 of which 20,683 with ETMs. Owen et al. only selected projects with records containing reserves and resources, as projects without reported reserves and resources are typically in early stage of development (before resource definition), have ceased to operate, or have not disclosed the information publicly. 964 of the 5097 geolocated mining projects intersect with Maus et al. (2022) dataset covering 49,576 km² of land used in 2019 by the global mining industry, including large-scale and artisanal and small-scale mining. We applied a 10-km buffer around project coordinates to account for the area of influence of mining sites and for the expansion of open cuts along the years (mining sites were found to cause deforestation as far as 50-km away (Giljum et al., 2022; Sonter et al., 2020)). Maus et al. (2022) state that a 10 km radius covers *‘most of the mining complexes spreading over several kilometers, including the largest mines in the world, which have an open cut extending over 4 km diameter’*.

Bioenergy (BE)

- **Pledges:** We used the net-zero scenario of the International Energy Agency (IEA) (IEA, 2021), which achieves net-zero (NZE) *‘with no offsets from outside the energy sector,*

and with low reliance on negative emissions technologies.⁵⁰ To limit conflicts with other land uses, notably food production, the NZE scenario constrains the BE production in 2050 to around 100 Exajoules (EJ) (i.e. 30% of the global energy production in 2050), with 10% used with CCS to capture 1.3 Gt CO₂ per year. 25 EJ of these 100 EJ come from short-rotation woody crops and 3 EJ from conventional BE crops (e.g. corn, soy, palm oil), i.e. 8% of the global energy production. This scenario is considered by IEA as a conservative sustainable BE potential (Creutzig, 2014), though recent studies tend to be more cautious (Creutzig et al., 2021; Kalt et al., 2020). The NZE scenario increases the total land area dedicated to bioenergy production to 410 Mha in 2050, with 270 Mha being forest (~1/4th of the total area of global managed forests, and ~5% of total forest area), 130 Mha used for short-rotation woody crops (i.e. lignocellulosic bioenergy) and 10 Mha for conventional BE (from ~75 Mha today). According to IEA, woody energy crops allow for greater production on marginal lands and can produce twice as much BE per ha compared to conventional BE. To the best of our knowledge, no global dataset of the BE area pledged in the energy sector (with and without carbon capture) is publicly available. We thus used the land gap report dataset which compiled the existing public BE pledges part of the NCDs and other climate pledges (Climate Resource, 2023) - with quantified values available only for five countries, estimated to be between 70-80 Mha in total. Conversion from emissions-based pledges to areas were based on country specific median yield values (Li et al., 2020), with conversion efficiency of 60% (Vaughan et al., 2018). We used the upper end of expected CDR of 80 Mha (with the USA accounting for 65 Mha), and we assumed that these pledges only targeted second generation bioenergy, thus leaving 50 Mha of IEA's total 130 Mha woody bioenergy to be distributed in other countries.

- **Current distribution:** We did not consider the current BE distribution, due to the lack of data and because the NZE scenario would lead to substantial distribution change.
- **Identification of targeted areas:**

For conventional BE, we used the BE DPI map of Oakleaf et al. (2019) produced based on the estimated yield for five major first-generation biofuel crops. The authors modelled the crop-specific relationships between area-weighted yield (ton/km²) and biophysical covariates (e.g., growing degree-day, precipitation, fraction irrigated, slope, etc.) to produce global, crop-specific, predicted yield maps. Lands classified as urban or currently cropped, arid lands without irrigation, and areas too steep for cultivation were removed. As the currently cropped land was removed from the DPI map, we assumed that the 10 Mha of conventional BE would be produced on new (potentially marginal) land.

⁵⁰ The NZE scenario is 'designed to maximise technical feasibility, cost-effectiveness and social acceptance while ensuring continued economic growth and secure energy supplies. [...] In the net-zero emissions pathway presented in this report, the world economy in 2030 is some 40% larger than today but uses 7% less energy. A major worldwide push to increase energy efficiency is an essential part of these efforts, resulting in the annual rate of energy intensity improvements averaging 4% to 2030 – about three-times the average rate achieved over the last two decades. Our pathway calls for scaling up solar and wind rapidly this decade, reaching annual additions of 630 gigawatts (GW) of solar photovoltaics (PV) and 390 GW of wind by 2030, four-times the record levels set in 2020. For solar PV, this is equivalent to installing the world's current largest solar park roughly every day. Hydropower and nuclear, the two largest sources of low-carbon electricity today, provide an essential foundation for transitions.'

For woody BE, we re-ran the DPI bioenergy code of Oakleaf et al. with Li et al. (2020) lignocellulosic BE yield map. The authors produced the global yield maps ($0.5^\circ \times 0.5^\circ$) of five important lignocellulosic bioenergy crops (miscanthus, switchgrass, eucalypt, poplar and willow) under current technology, climate and atmospheric CO₂ conditions, using a random-forest algorithm to upscale a bioenergy yield dataset of 3963 observations using climatic and soil conditions as explanatory variables. We used the combined 'best bioenergy crop' yield map (selecting the yield of the crop type with the highest yield in each of the grid cells, eucalypt and Miscanthus in most cases). We updated the constraint mask in order not to mask the currently cropped land (this could not be done for the conventional BE as the yield map was already initially masked), assuming that these second generation bioenergy crops could replace current crops. We do not spatialize the 270 Mha of bioenergy from forestry, though we included it in the expected overlaps with the current forestry land use.

- **Selection:** Of these 140 Mha of bioenergy crop, the NZE scenario allocates 70 Mha of marginal lands or land currently used for livestock grazing and 70 Mha of cropland to bioenergy. We did not follow this assumption, as there is no formal definition of 'marginal lands' or regulatory scheme currently constraining bioenergy crops on these lands, and thus, in line with the other interventions and under market forces, we consider that bioenergy production would likely target lands that have the highest potential for production.

Text S1.4. Current major land uses

To capture potential competition with current major land uses, we used a recent global land use dataset that traces global land-use area and intensity developments between 1992 and 2020 annually at 30 arcsecond spatial resolution (Matej et al., 2025). We used the aggregated classes provided in separate maps; cropland, grazing land, forestry, built-up land and wilderness (which we renamed 'area with minimal land use'). For grazing land, we kept the two distinct classes: (i) 'no trees', defined as grazing areas with high suitability, thus named here 'more intensive', (ii) 'open wooded land', which can be harvested but with a lower human influence than in the first grazing class, named here 'less intensive'. For 'wilderness' we kept the distinction between snow and deserts ('unproductive' land), and other 'productive' wilderness areas defined based on the Net Primary Production. To align these maps with the other datasets, we first converted the absolute km² area given per cell to fractional cover per cell (by dividing the absolute area by the cell area), then reprojected them to Mollweide with 1x1 km resolution using the bilinear method for continuous data.

Once merged, 64% of the cells had more than 100% cover (summed areas of all classes exceeding 1 km²), though this represented only 0.4% of the total area. We adjusted the area of each land use to reach a maximum of 1 km² cover per pixel. We started by reducing the area of the wilderness class in these cells (by subtracting the area over 1 to the wilderness class), dropping the number of cells with >100% cover to 42%, then we did the same for grazing land (6%), forestry (0.5%), and built-up areas (0%). The total area of each class was reduced by less than 1.5% of their area, and the total area was reduced by 0.4%.

Supplementary tables

Table S1. Business-as-usual land demand estimates by 2050 from the literature. Values for 2020-2050 are reported in the last two columns; (i) based on the current land use areas as per Matej et al. (2025) to which we applied the annual percentage of increase (%/y), or (ii) based on the annual area increase (Mha/y) for 30 years - depending on data availability. Red values in brackets are considered as outliers and are not included in our range. Green (blue) values are kept for the low (high) estimate, marked with one star (two stars). For estimates based on SSPs, we only kept SSP2 considered as a continuation of current trends.

Reference	Low/ High	Year1	Year2	Year1 (Mha)	Year2 (Mha)	Increase (Mha)	Increase (%)	Increase (%/y)	Increase (Mha/y)	(i) Based on Matej + % increase	(ii) Based on Mha 20-50
CROPLAND											
(Matej et al., 2025)		2020		1,570							
(Bajželj et al., 2014) / Baseline CT1 scenario		2009	2050	1,560	2,220	660	0.42	0.010	16.10	(486)	(483)
(Alexandratos & Bruinsma, 2012)		2005	2050			69	0.047	0.001	1.53	49	46*
(FAO, 2018)		2012	2050	1,567	1,732	165	0.11	0.003	4.34	131	130
(Hertel et al., 2016) in (Meyfroidt, 2018)									4.00		120
(Hertel et al., 2016) / fig2		2006	2050			310	0.20	0.004	6.89	209	207
(Rogelj et al., 2018) / Baseline SSP2, fig.2f (approximate value)	Low	2010	2100			75			0.83		(25)
	High	2010	2100			280			3.11		93
(Schmitz et al., 2014)	Low	2005	2050				0.10	0.002		105	
	High	2005	2050				0.25	0.006		262	
(Searchinger et al., 2023)		2010	2050			200			5.00		150
(Stehfest et al., 2019) / fig.1, SSP2	Low	2010	2050	1,200	1,300	100	0.08	0.002	2.50	98	75
	High	2010	2050	1,200	1,500	300	0.25	0.006	7.50	294	225
(Tilman & Clark, 2014) / fig.4c, Income- dependent diets		2009	2050			600			14.63		(439)
(Weindl et al., 2024) ¹		2020	2050	1,616	1,742	126	0.08	0.003	4.22	123	126
(Weindl et al., 2024) ²		2020	2050	1,650	1,919	270	0.16	0.005	9.00	257	270**
(Willett et al., 2019) / BAU, fig6		2010	2050	1,260	2,110	850	0.67	0.017	21.25	(794)	(638)
GRAZING LAND											
(Matej et al., 2025)		2020		4,710							
(Bajželj et al., 2014) / Baseline CT1 scenario		2009	2050	3,280	3,710	430	0.13	0.003	10.49	452	315**
(Searchinger et al., 2023)		2010	2050			400			10		300
(Rogelj et al., 2018) / Baseline SSP2, fig.S19c (approximate value)	Low	2010	2100			-200			-2.22		(-67)
	High	2010	2100			200			2.22		67
(Weindl et al., 2024) ¹		2020	2050	3,289	3,331	42	0.01	0.000	1.40	0	42*
(Weindl et al., 2024) ²		2020	2050	3,267	3,343	76	0.02	0.001	2.53	94	76

Appendix Chapter 4

BUILT-UP LAND											
(Matej et al., 2025)				160							
(G. Chen et al., 2020)	Low	2010	2050	60	97	37	0.62	0.015	0.93	74	28*
	High	2010	2050	60	108	48	0.80	0.020	1.20	96	36
(Fischer et al., 2012) in (Searchinger et al., 2023)		2000	2050	152	233	81	0.53	0.011	1.62	51	49
(Huang et al., 2019) / SSP2 from fig2a (approximate value)	Mid (SSP2)	2015	2050			110			3.14		94
(Searchinger et al., 2023)		2010	2050			80			2.00	0	60
(Simkin, 2022)	Low	2015	2050			82	0.59	0.017	2.34	80	70
	High	2015	2050			153	0.59	0.017	4.37	80	131**
(Weindl et al., 2024) ¹		2020	2050	61	94	34	0.56	0.019	1.13	89	34
(Weindl et al., 2024) ²		2020	2050	74	111	36	0.49	0.016	1.21	78	36
(Zhou et al., 2019)		2012	2050	127	173	46			1.22		37
FORESTRY FOR TIMBER											
(FAO, 2022) / Planted forests (Industrial roundwood only, excluding BE)	Low	2020	2050	294		70			2.33		70
	High	2020	2050			172			5.73		172**
(Mishra et al., 2021) / not taken in account because only plantations		1995	2100			171			1.63		(49)
(Nepal et al., 2019) / Planted forests	Low	2015	2070	285	379	94			1.71		51
	High	2015	2070	285	475	190			3.45		104
(Searchinger et al., 2023) / Plantations (excluding BE)	Low	2010	2050	203	271	68			1.70		51*
	High	2010	2050	203	556	353			8.83		(265)

¹SHAPE REMIND-MagPIE, SSP2-NPi (baseline)

²SHAPE IMAGE, SSP2-NPi (baseline)

Table S2. Renewable energy installed capacity density factors from the literature. Only estimates of *installed* capacity were retained, and only the space footprint of onshore wind turbines. Red values in brackets are considered as outliers and are not included in our range. Orange values marked with two stars are kept for the mid estimate used for the spatialisation of targets. Green and blue values marked with one star and three stars are used for the range of low and high estimates, respectively.

Reference	Initial Unit				km2/MW		
	Unit	Mid	Low	High	Mid	Low	High
SOLAR (Utility-scale PV)							
(Bolinger & Bolinger, 2022)	MW/acre	0.35	0.50	0.25	0.012**	0.008*	0.016
(Y. Chen et al., 2022)	m2/MW	18,000					0.018
(Jacobson et al., 2022)	km2/MW				0.012**		
(Ong et al., 2013) in (Trainor et al., 2016)	km2/GW	34.40	28.20	38.90	0.034	0.028	0.039***
(NREL, 2009) in (King et al., 2023)	MW/km2	3.00			0.030		
(Tröndle, 2020) (range from multiple studies)	W/m2	70.00	100.00	20.00	0.014	0.010	(0.050)
ONSHOREWIND							
(Y. Chen et al., 2022) based on (Enevoldsen & Jacobson, 2021) / total disturbed area (account for noise and visual impact)	m2/MW	200,000			0.20**		
(Enevoldsen & Jacobson, 2021) (Europe)	MW/km2	19.80	46.90	6.20	0.05*	(0.021)	0.161
(Enevoldsen & Jacobson, 2021) (outside Europe)	MW/km2	20.50	48.00	16.50	0.05*	(0.021)	0.061
(Jacobson et al., 2017) (SI p40)	W/m2	7.10			0.14		
(Jacobson et al., 2022)	km2/MW				(0.029)		
(Ong et al., 2013) in (Trainor et al., 2016)	km2/GW	202.30	368.30	465.4	0.20**	0.37***	(0.47)
(MacDonald et al., 2016) (SI page 18)	W/m2		3.22	2.10		0.31	(0.48)
(Miller & Keith, 2018) in (Johnson et al., 2021)	MW/km2	2.70			0.37		
(NREL, 2009) (p10) in (Stevens et al., 2017)	rotor diameter lengths	7.00	5.00	10.00	0.21	0.15	0.30
	rotor diameter lengths	15.00			0.45		
(Reichenberg et al., 2022) in (Hedenus et al., 2022)	MW/km2	5.00			0.20**		
(Tröndle, 2020) (range from multiple studies)	W/m2	10.00	19.00	2.00	0.10	0.05*	(0.50)
(van Zalk & Behrens, 2018)	W/m2	8.80	10.78	6.82	0.11	0.09	0.15
(Wu et al., 2017) in (Kiesecker et al., 2024)	MW/km2	5.00			0.20**		
CSP							
(Alami et al., 2023) / Solar field only	m2/MW		8,504	11,477		0.01	0.01
(Jacobson et al., 2022)	km2/MW				0.029**		
(Thonig et al., 2023) calculated based on the database (n = 92) / Total area (Solar field + plant area)	km2/MW				0.054***		
(Ong et al., 2013) / Total area	acres/MW	10.00			0.040		
(Ong et al., 2013) / Linear fershnel total area	acres/MW	4.70			0.020*		

Supplementary figures

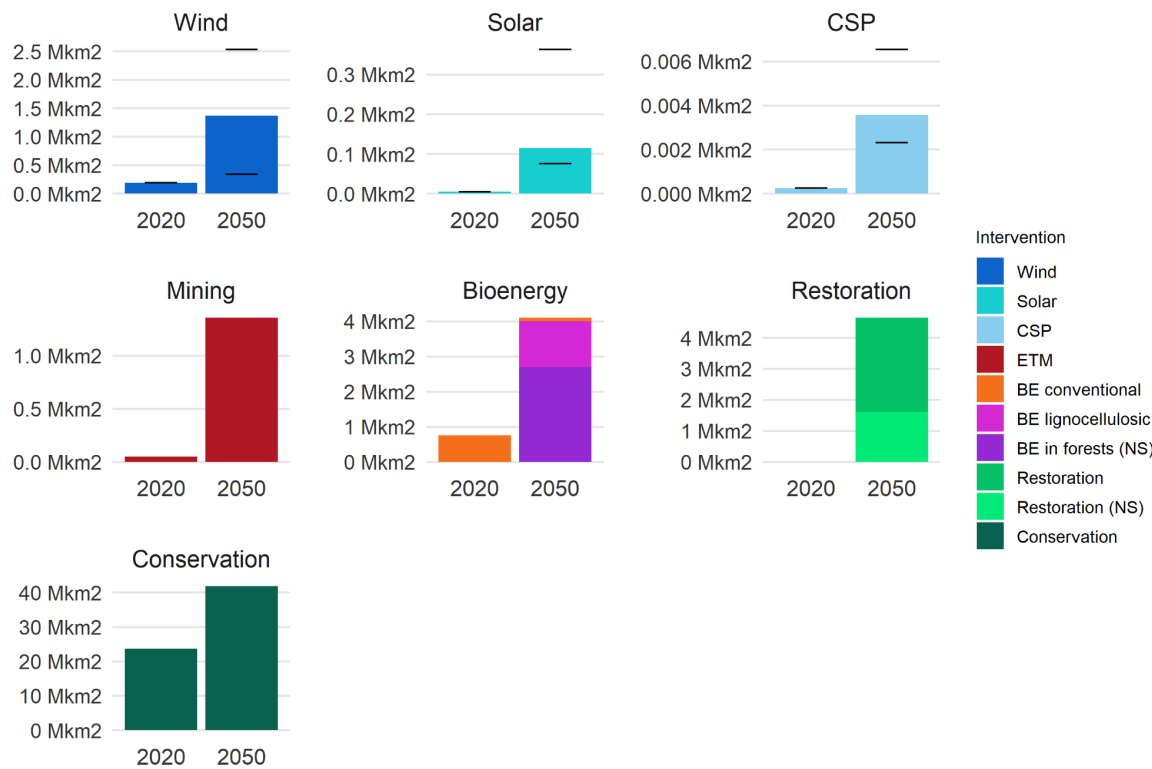


Figure S1. Absolute increase per intervention. ‘2020’ refers to current areas but the displayed areas do not all correspond to 2020 exactly. Horizontal lines for Wind, Solar and CSP correspond to the low and high estimates (cf. table 1).

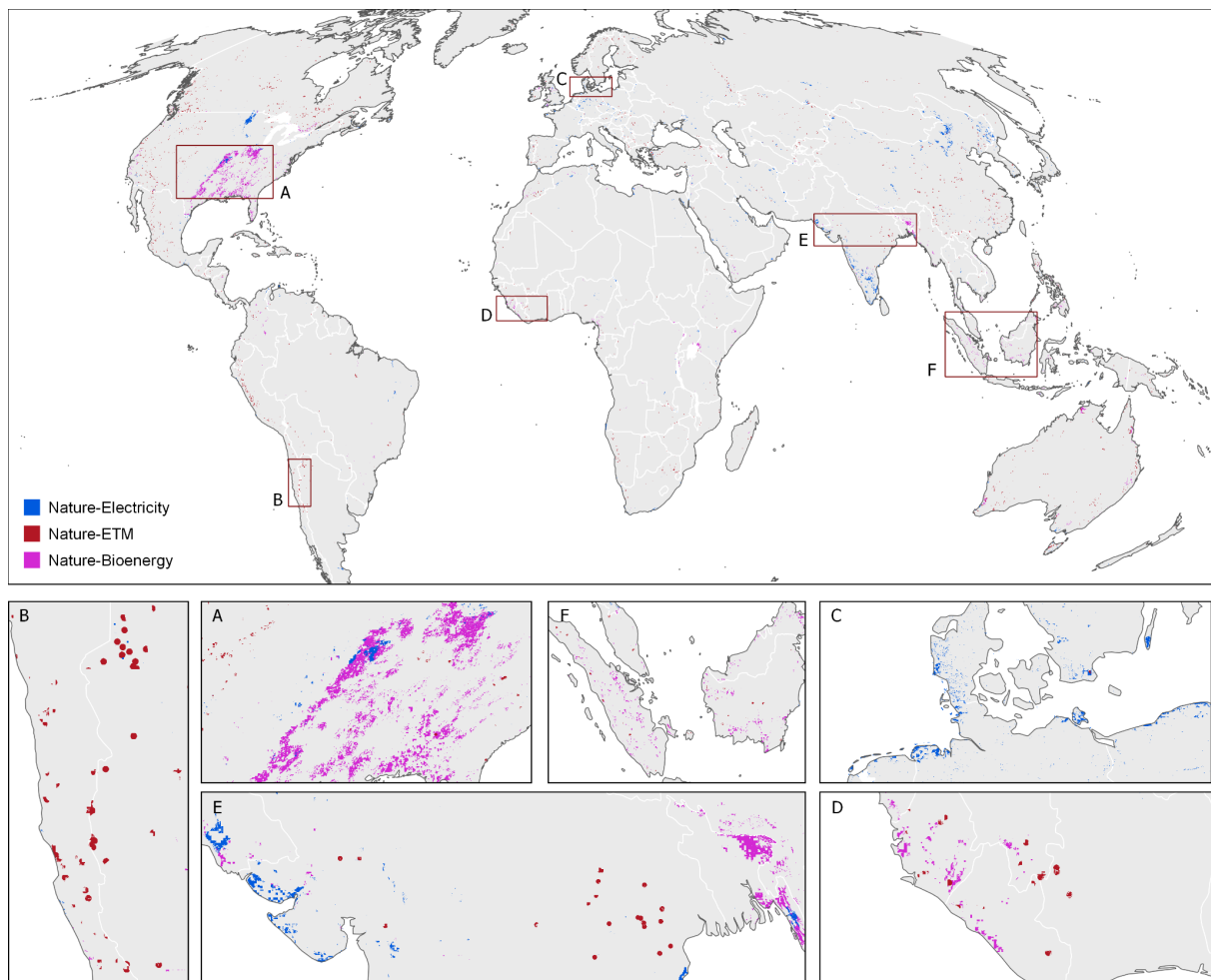


Figure S2. Map of overlaps between 2050 spatialised interventions (current and projection combined), with 'Electricity' gathering Wind, Solar, CSP, and 'Nature' gathering Restoration and Conservation. Maps were overlaid as shown in the legend (Nature-Bioenergy below, and Nature-Electricity on top).

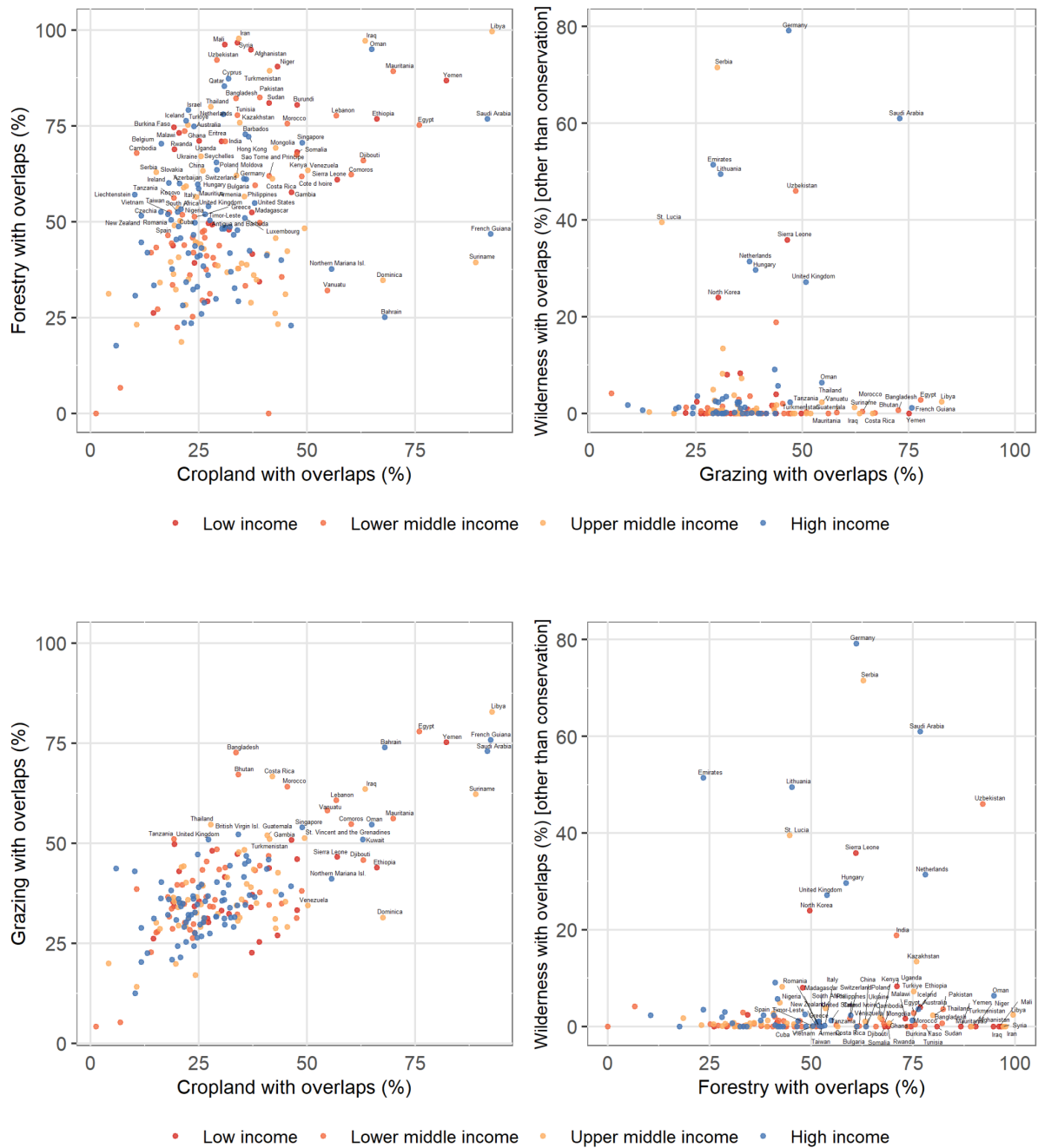


Figure S3. Percentage by country of current land covers overlapping with 2050 interventions. Percentage of the current area of cropland and forest (left) and of cropland and grassland (right) with at least one overlapping intervention in 2050, per country and dependent territories (>100 km² in area), colored according to their income group following 2024 World Bank's classification.

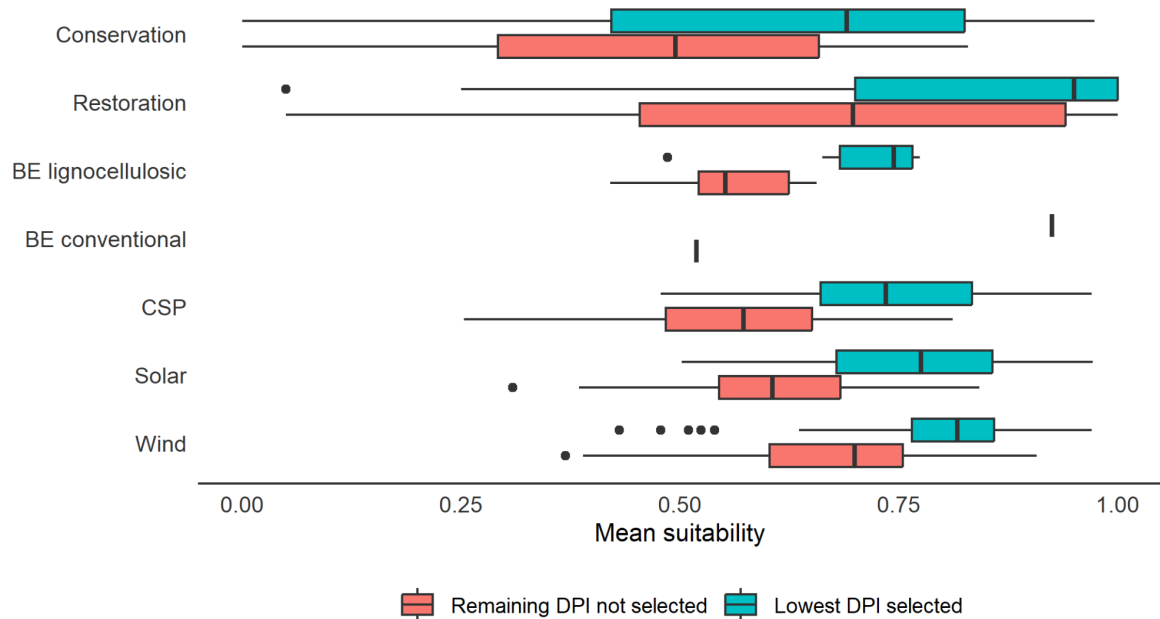


Figure S4. Suitability of selected pixels per intervention. In green, the average suitability of the last selected percentile in each country, thus representing the lowest suitability value required to fill in the pledge or target area in each country. In red, the average suitability of the area which is not targeted by any intervention in 2050 (as per our analysis), not including areas considered as not suitable. The boxplots represent the distribution of the countries' averages.

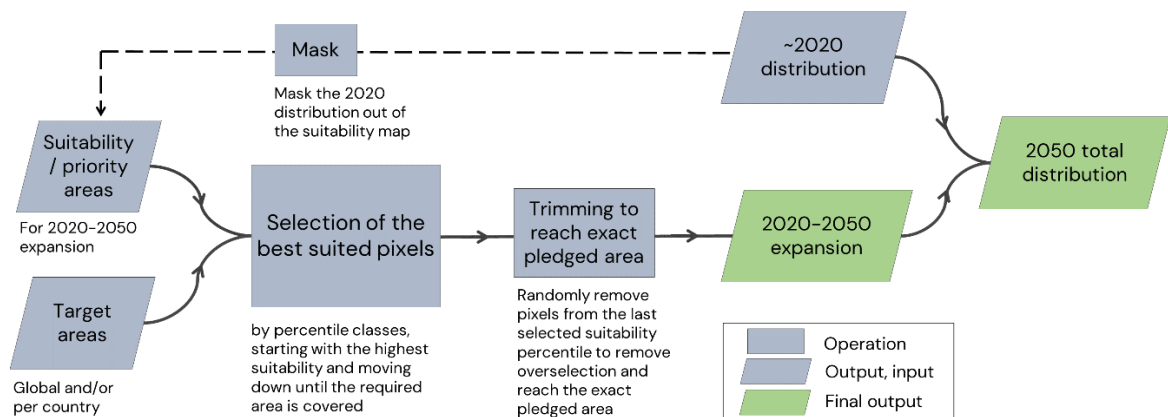


Figure S5. Workflow. In large countries the selection of the best-suited percentile can over-allocate land to each intervention - hence we trim these pixels to match the exact pledged area.

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