If you are going to lean on a tree: Beyond carbon-centric climate mitigation

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If you want to lean on a tree, first make sure it can hold you. – Ambede proverb

This thesis is dedicated to

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for being the shoulders on which I could stand on

Abstract

Anthropogenic climate change increases risks to social and ecological systems, necessitating mitigation measures. At present, mitigation is predominantly organised through the paradigm of net zero, with a concomitant metric of success being the quantity of carbon (and other greenhouse gases) reduced or sequestered. However, this metric may obscure the wider context in which a mitigation effort is situated. This thesis is interested in three examples of where this occurs: mitigation's interactions with other climate interventions; ecological constraints and trade-offs; and socio-economic consequences. Thus, a novel typology is first presented, which situates mitigation activities within a broader system of climate risk management interventions, including adaptation measures and other mitigation efforts. This allows for the identification of feedback loops and potentially impactful interactions. Second, the extent to which current mitigation activities are well-situated within their host ecological system is assessed, by quantifying the extent to which forestation projects are sited in areas that would naturally be dominated by closed-canopy forest ecosystems. The results show that a majority are not, suggesting that carbon-centrism may be disconnecting mitigation from its ecological context. Finally, urban nature-based solutions in Johannesburg, South Africa, are evaluated relative to the wider socio-ecological context of the city. The findings indicate that the native grassland ecosystem could be as effective as the novel urban canopy at providing temperature and biodiversity benefits, although this may be more related to the location of the ecosystem. Nonetheless, the respective temperature benefits are particularly important since lower income households were found to correlate with higher heat outcomes. Altogether, these results demonstrate that a carbon-centric approach to mitigation may miss critical context that can inform how just, ecologically sound, and thus durable, mitigation interventions are. Awareness of the wider (socio-ecological) context is therefore essential for safe-guarding the efficacy of mitigation efforts over a decades-long transition to achieving our climate goals.

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Abbreviations

AMOC	Atlantic meridional overturning circulation
AR6	IPCC Sixth Assessment Report
BII	Biodiversity intactness index
CDR	Carbon dioxide removal
CO_2	Carbon dioxide
FAO	Food and Agriculture Organization
GDP	Gross domestic product
GLM	Generalized linear model
IAM	Integrated assessment model
IPCC	Intergovernmental Panel on Climate Change
LST	Land surface temperature
MODIS	Moderate Resolution Imaging Spectroradiometer
NbS	Nature-based Solutions
NDC	Nationally determined contribution
PCA	Principal component analysis
SANLC	South African National Land Cover
SES	Socio-ecological system
SRM	Solar radiation modification
UN	United Nations
UNFCCC	United Nations Framework Convention on Climate Change
WGI/II/III	IPCC working group 1/2/3

1. Introduction

"The Paris Agreement is a monumental triumph for people and our planet." – United Nations Secretary-General Ban Ki-moon

"The pursuit of a net-zero target is perhaps the most ambitious collective undertaking in human history." – World Resources Institute

Figure 1: *The 'monumental triumph' of reaching agreement in Paris.*

The photograph of Laurence Tubiana, Christiana Figueres, Ban Ki-moon, Laurent Fabius, and Francois Hollande, hands aloft and smiles wide, has entered climate lore as a representation of the 'monumental triumph' of reaching agreement in Paris (**Figure 1**) (UN News Centre, 2015). The emblematic nature of the image, of the moment, has some justification: the legally binding framework emerged from years of complex, near-deadlock negotiations with ambitions almost-miraculously intact (Rajamani, 2016; Streck et al., 2016). The resulting Paris Agreement struck a fine political balance between reducing climate-related risks and the costs of decarbonising the global system at pace and, in so doing, embedded net zero into the international zeitgeist (Green et al., 2025; Rajamani, 2016).

The genius of net zero as a framing for climate mitigation is its simplicity: it requires no more and no less than the balancing of the scales. To wit, by mid-century any anthropogenic carbon dioxide emission into the atmosphere must be matched by an anthropogenic removal of the same quantity (Fankhauser et al., 2022), departing from a previous mitigation focus on achieving a specific greenhouse gas concentration. Part of the appeal of net zero is the relative tidiness of the framing: the goal is clear (a balance between emissions and removals), the dates are relatively certain (mid-century or thereabouts), and the messy business of operationalising decarbonisation is left to the entity in question (Welton, 2022). This pristine minimalism means that the concept has metamorphosed from the relative obscurity of physical climate science to become the organising framework for climate mitigation (Allen et al., 2022; Welton, 2022).

Net zero's inherent balancing act means that climate mitigation is often framed and implemented in terms of "pristine carbon balance sheets" (Welton, 2022, p. 176), with interventions thus defined in terms of a quantifiable emissions reduction (Burley Farr et al., 2023; Lee et al., 2024). In other words, carbon has become the dominant metric through which mitigation is understood, compared, and governed.[1] This has its benefits, to be sure. The clearest is its simplicity. It also permits comparison, quantification, and standardisation, with carbon metrics enabling the consistent measurement and comparison of mitigation efforts across actors and contexts (Stachelscheid & Dutzi, 2025). Carbon thus renders the complex challenge of climate change governable.

[1] This thesis is particularly concerned with carbon dioxide, rather than a comprehensive interrogation of the full suite of greenhouse gases. This is both due to carbon's importance in global mitigation scenarios and to ease of explanation. The nuances differ for each individual greenhouse gas, but many of the same issues canvassed here also apply.

However, the translation of net zero (and its focus on carbon) to the complexity of the human enterprise tests the limits of its elegant simplicity. Indeed, while the achievement of reaching consensus in Paris was monumental, reaching and maintaining net zero will be exponentially so – requiring concerted and co-ordinated action across time, geographies, institutions, and ideologies (Hale, 2024; Meadowcroft & Rosenbloom, 2023). This translation raises questions of equity, durability, and political prioritisation across a wider backdrop of policy interests (Lenzi et al., 2021; Meadowcroft & Rosenbloom, 2023; Zickfeld et al., 2023). In other words, the simplicity that makes carbon such a powerful metric may also obscure some important weaknesses.

As such, a critical question is how resilient mitigation is to the 'inconvenient complex unruliness' of the context within which it is embedded (Stirling, 2026, p. 3). Each mitigation intervention must enter into, or alter, existing social and ecological systems. These systems are characterised by ecological constraints, competing policy objectives, and existing patterns of inequality (Markkanen & Anger-Kraavi, 2019; Searchinger et al., 2023). As a result, interventions designed primarily around carbon outcomes may generate wider governance, ecological, and social consequences that are not captured by carbon accounting alone (Carton et al., 2021; Lövbrand & Stripple, 2011). This can impact wider processes (including other climate risk management interventions), as well as the mitigation effort itself.

This tension is particularly apparent in land-based mitigation, which is premised upon enhancing the natural carbon cycle so as to sequester greater quantities of CO_2. Land-based mitigation projects are popular components of net zero strategies, given their relative cost efficiency (Doelman et al., 2020; S. Smith et al., 2024). This appeal is bolstered by projects being pitched as a 'triple win' with biodiversity and social co-benefits (Girardin et al., 2021; Muradian et al., 2013). An especially popular land-based mitigation technique is reforestation or reforestation (together ‘forestation’), whether in large-scale projects or in an urban context (Bherwani et al., 2024; S. Smith et al., 2024).[2]

However, mitigation measures like reforestation and afforestation do not exist within a vacuum. Instead, they must interact with competing policy concerns, including other climate efforts (Azevedo & Leal, 2017). They also engage with local ecological conditions and processes like water availability or disturbance regimes, and social factors such as historic patterns of land use, access, and inequality (Berkes et al., 2000; Bodin & Tengö, 2012; Luedeling et al., 2019; Sekulova et al., 2021).

While these interactions may generate co-benefits (Key et al., 2022; Seddon, 2022), they also risk misplacement of efforts in inappropriate ecosystems or uneven distributions of costs and benefits (Baldwin-Cantello et al., 2023; Enevoldsen et al., 2022; Pradhan et al., 2017; Weiskopf et al., 2024). Forestation located in ecologically inappropriate areas can undermine biodiversity, water availability, and may even threaten existing carbon stores (Briske et al., 2024; Honda & Durigan, 2016; L. Li et al., 2023; Prangel et al., 2023; Ullah et al., 2025). It may additionally impact local livelihoods, access to land, and the provision of ecosystem services, with uneven consequences for different groups (Luvuno et al., 2022; Prangel et al., 2023; Skhosana et al., 2025). Such risks are non-trivial, as they may impact the durability of the mitigation effort by reducing its long-term ecological or social viability, which in turn can delay the achievement of net zero. Further, they may impact the ability of the mitigation measure, as well as the host ecosystem and communities, to adapt to the rising risks related to climate change (Buck et al., 2020; Rodriguez Morales, 2025; Work et al., 2019). For example, reduced water availability resulting from ecologically misplaced

[2] In this thesis, reforestation broadly refers to tree planting in areas that previously supported tree-dominated ecosystems, while afforestation is in areas that have long been non-tree-dominated.

forestation may leave the project itself less resilient to future droughts and heatwaves, while also reducing water resources available to surrounding ecosystems and communities.

Accordingly, a number of stakeholders have begun to call for more integrative and systemic approaches to mitigation (Bravo et al., 2026; Council for Science and Technology, 2020; Ison & Straw, 2020). For example, the nature-based solutions framework specifically recognises the interlinkages between climate change, nature, and communities (Seddon et al., 2021), while the systems thinking perspective is increasingly being incorporated into the mitigation literature and practice (Bravo et al., 2026; Geels et al., 2017; Ison & Straw, 2020; Qudrat-Ullah, 2025). These perspectives proceed from the basis that mitigation should not be understood solely in terms of carbon but must also account for how individual interventions interact with the relevant ecosystem dynamics, governance structures, and community concerns.

However, these wider perspectives are not yet consistently reflected in the design and evaluation of mitigation efforts (Howarth & Robinson, 2024; Markkanen & Anger-Kraavi, 2019; Strauß et al., 2022); the incorporation of broader socio-ecological considerations is often partial or case-specific (Lah, 2025; Pettorelli et al., 2021). Climate mitigation and adaptation, alongside other socio-ecological priorities such as biodiversity conservation and human well-being, are often considered or implemented by different actors, at different scales, and through different analytical frameworks (Pettorelli et al., 2021; Singh & Chudasama, 2021; Zhao et al., 2018). While integrated approaches have been partially incorporated into mitigation, significant gaps remain. Accordingly, new conceptual approaches and empirical analyses that situate mitigation within relevant social, ecological, and governance systems are required.

This thesis is intended as a response to that observed research gap. To do so, it situates climate mitigation within intertwined social, ecological, and governance contexts. This doctorate argues that understanding mitigation through this lens is vital for identifying the conditions under which interventions are likely to be effective, durable, and equitable, and for recognising the consequences of approaches that remain fragmented. Combining a conceptual typology and two empirical analyses at a global and local scale, the thesis considers how mitigation interacts with other climate interventions, ecological contexts and social outcomes, and the resulting implications for mitigation design. In so doing, it demonstrates how approaches that focus too narrowly on carbon can overlook important institutional interactions, ecological constraints and social consequences, and how an integrated perspective can help make these factors more visible. This is critical for ensuring the durability of individual climate mitigation measures and thus the entire net zero endeavour itself.

1.1. Aims and research questions

The aim of this thesis is to examine the effects of the current carbon-centric approaches to mitigation and to consider alternative, integrated approaches, using a combination of conceptual and empirical chapters. It rests upon the supposition that mitigation cannot be understood solely in terms of carbon but should also account for how interventions interact with ecological systems and social processes.

The thesis approaches this aim in two ways. First, it develops a framework for positioning mitigation within its broader governance context, with reference to other mechanisms that have been put forward to decrease climate risk, such as adaptation or solar radiation management. Second, it presents two empirical case studies (at a global and local scale, respectively). These examine how mitigation interventions are situated within socio-ecological contexts, and the potential consequences for mitigation design. The first empirical study is focused on global forestation efforts and aims to understand the extent to which these projects are aligned with the relevant ecological context. The second empirical paper concerns urban forestation efforts in a city that is situated within an ecosystem in which trees are not naturally present. It aims to compare the benefits that accrue from a novel forest ecosystem, as compared to the native grassy ecosystem and the more built-up urban core. Such a comparison aims to present an example of how to approach mitigation with a wider lens than a pure focus on carbon.

The research questions that this study aims to answer are:

RQ1 (Paper 1): How can climate mitigation and other climate risk management strategies be conceptualised in a way that makes their trade-offs and synergies more visible to decision makers and researchers?

RQ2 (Paper 2): To what extent are afforestation and reforestation projects for climate mitigation occurring in ecologically appropriate areas?

RQ3 (Paper 3): How do novel, native, and urban-industrial ecosystems differ in the climate, biodiversity, and human well-being benefits they provide as urban nature-based solutions, and are these benefits distributed equitably across race and income in Johannesburg?

These three research questions aim to consider from different angles the extent to which carbon-centric approaches may miss wider systemic dynamics and whether integrated approaches might provide a fruitful alternative.

1.2. Thesis outline

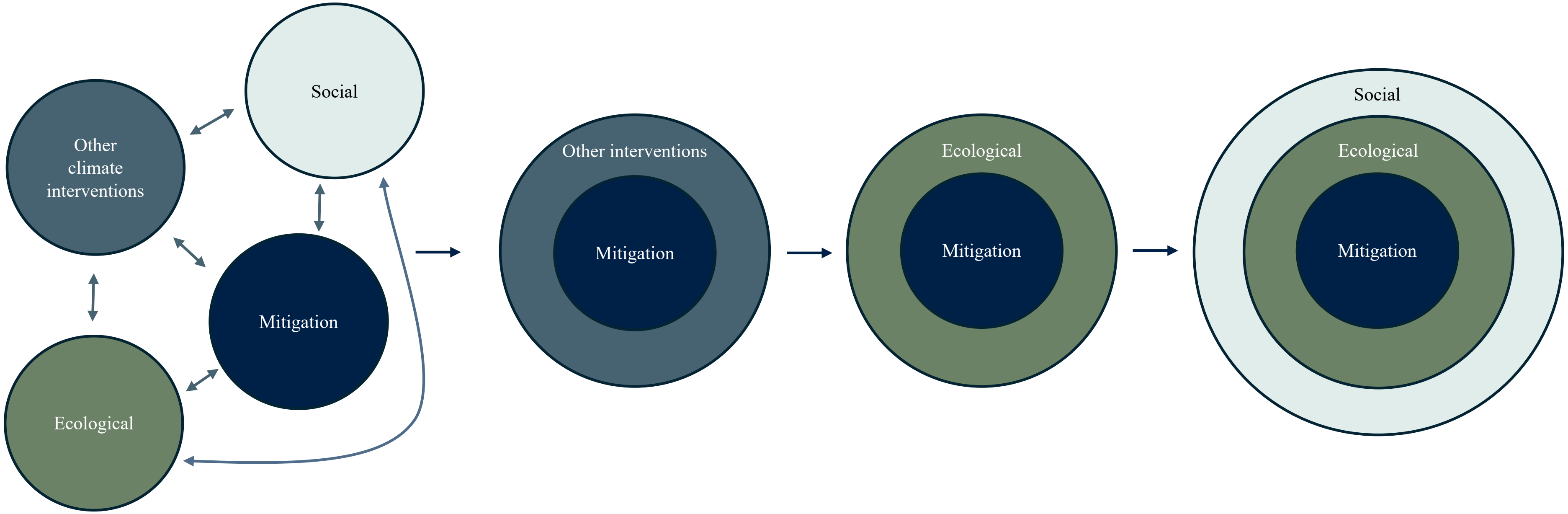
This thesis follows the structure and requirements for an article-based thesis. Chapters 1-2 provide context for the three research papers (chapters 3-5), which, in turn, provide the primary academic contribution of this thesis (**Figure 2**). Chapters 3 and 5 have been published in international, peer-reviewed academic journals, while Chapter 4 has been prepared for submission to the same. Chapter 6 synthesises the learnings from the three papers and concludes the dissertation.

This chapter has presented the broad background to the research aim and questions with which this dissertation is concerned. **Chapter 2** reviews the literature that informs the approach in this thesis, identifying the gaps that this dissertation aims to fill. The chapter focuses on two key themes. First, it outlines how carbon has become the dominant metric around which mitigation efforts are organised. This is followed by a consideration of how such a focus may miss wider impacts on other elements within the relevant ecological, social, and governance systems.

Chapter 3 (paper 1) builds on this introduction through inductive review and presents a new conceptual framework for integration of all climate interventions (including mitigation) into one risk management typology. The typology rests on two assumptions: that each intervention should ideally work to decrease risk, and that a consideration of all interventions together can promote the identification of synergies and trade-offs, so as to more effectively decrease risk overall. To illustrate the utility of the typology, the chapter also presents six stylised portfolios of climate scenarios, which consider how various interventions may decrease risk in varying mitigation scenarios.

Paper 2 (**chapter 4**) presents a global empirical case study of the extent to which real-world climate mitigation projects have considered the local ecological context. The study identifies areas where verified afforestation and reforestation projects are sited in areas that do not ecologically support forests. Half a million forestation planting site records are overlaid with spatial maps of three complementary ecological frameworks for identifying areas suitable for forests (i.e. ecoregions, predicted natural tree cover, and biophysical forest potential).

After the previous chapter's recognition that the ecological alignment is critical, paper 3 (**chapter 5**) adds an additional consideration, social context, to reflect the entanglement between social and ecological systems. The aim of this chapter is to identify whether native or novel ecosystems might be appropriate foundations for nature-based solutions in urban



Current landscape. Climate mitigation is often approached as a discrete, carbon-focused activity, despite being embedded within broader climate risk management, social and ecological systems

Paper 1. A Climate Risk Management Typology

conceptual

Paper 2. Widespread misplacement of forestation projects in non-forest ecosystems

empirical: global

Paper 3. Comparing the benefits of native and novel ecosystems as urban nature-based solutions: a case study of Johannesburg, South Africa

empirical: local

***Figure 2**: **Conceptual structure of the DPhil thesis.** The foundation of the dissertation is that the current approach to climate mitigation is carbon centric and thus often siloed from other adjacent concerns and a broader context. This thesis responds by developing the integration of mitigation to a wider context (other climate interventions, ecological systems, and social concerns) through (i) the development of a climate risk management typology that conceptually integrates mitigation with other climate risk management interventions (Paper 1; Chapter 3); (ii) empirically illustrating a failure to procedurally integrate ecological context into climate mitigation action at a global level (Paper 2; Chapter 4); and an empirical case study of how biodiversity and human well-being can be integrated into climate policy at a local level (Paper 3; Chapter 5).*

areas, and how this is distributed across demographic and economic lines. To achieve this, the paper uses an empirical case study of Johannesburg, South Africa, to consider, at a local level, the climate, biodiversity, and human well-being benefits of the native grassy ecosystem and the novel wooded ecosystem, as compared to the urban-industrial areas of the city. We combine MODIS land surface temperature, a biodiversity intactness index, and national census datasets at a ward-level to run regression models. This process provides a blueprint for situating mitigation actions within their socio-ecological context.

Last, **chapter 6** synthesises the three papers to draw out learnings and potential future directions that emerge from the thesis. Ultimately, the thesis argues that climate mitigation policy, research, and action require an appreciation for the fact that each mitigation intervention operates within a specific ecological and social context. Moving beyond carbon centrism to view the interaction between the nodes in the system can go some way to having a more efficient and robust response to climate change.

A GenAI use declaration, as well as the supplementary materials to **chapters 4** and **5**, are included in the **appendices**. Further, **chapter 3** was published in an 'article forum' in *Dialogues on Climate Change*, which solicits responses to each article published. After receiving two such responses, we were asked to respond in kind. Our response, which further explores some of the ideas presented in the initial paper, is also included in the **appendices**.

The three research papers of the thesis work together to advance the understanding of the necessity of situating mitigation efforts within the context of the wider, governance, social, and ecological context. As set out in **Figure 2**, the first paper builds a conceptual framework for integration; the second tests the extent of integration in a forestation context; the third demonstrates its application in an urban socio-ecological context. By linking conceptual and empirical research across scales, the thesis provides a holistic analysis of how situating mitigation with the relevant context can strengthen its effectiveness, durability, and potential for equitable outcomes.

2. Literature review

> *"… we see repeated the same basic paradoxes: man, the conqueror versus man the biotic citizen; science the sharpener of his sword versus science the search-light on his universe; land the slave and servant versus land the collective organism."*
>
> – Aldo Leopold, *A Sand County Almanac*

This chapter addresses the wider background and relevant literature that frame the thesis as a whole. With this in mind, only a subset of a much larger literature is presented, with further literature reviews relevant to each chapter discussed in that context.

This chapter argues that the current framing of mitigation that has its roots in the UNFCCC and that was further developed by the Paris Agreement incentivises purely carbon-focused approaches. This is part of an effort to render climate change 'governable'; however, this impulse risks losing sight of potentially meaningful interactions with the broader system. Integrated approaches are an emerging response. Consequently, this chapter argues for a wider view of mitigation, moving beyond carbon accounting alone to consider how mitigation interventions interact with broader ecological, social, and governance contexts. In so doing, it puts climate science and global governance literatures into conversation with critical social theory, and ecological considerations.

2.1. Making climate change governable

The most recent set of reports from the Intergovernmental Panel on Climate Change (IPCC) reaffirmed a well-established finding: anthropogenic greenhouse gas emissions are affecting our climate, and this will have increasing impacts on human societies and ecosystems (IPCC, 2021, 2022b, 2023b). Climate and weather extremes such as floods, droughts, extreme heat, and increased wild fires have already been observed and attributed to climate change (Barnes et al., 2025; Clarke et al., 2026; IPCC, 2022b; Kimutai et al., 2025). These events have implications for human health, biodiversity, food systems, infrastructure, and livelihoods, amongst others. Continued emissions increase the risks of hazards and impacts in the future (IPCC, 2022b).

Humanity has recognised that this requires an urgent response (Kyoto Protocol to the United Nations Framework Convention on Climate Change, 1997; Paris Agreement to the United Nations Framework Convention on Climate Change, 2015; UNFCCC, 1992).[3] However, responding to climate change is a 'wicked' business, characterised by multiple "open, complex, and imperfectly understood systems" that interact across scales (Hulme, 2009, p. 334; Levin et al., 2012; Rittel & Webber, 1973). To govern such an issue requires, as a preliminary step, the rendering of it into a form that is legible to policymakers.

The process of rendering complex phenomena legible for governance has a long tradition. Such simplifications are useful for reducing complex realities to metrics that make them more easily recorded, monitored, and, not least, managed (Scott, 1998). These 'ways of seeing' incorporate a wide range of "practices, conventions, techniques, tools, infrastructures, and institutions that allow certain phenomena to be organized and understood as coherent objects of governance" (Boyd, 2010, p. 847). This process is alluring since it renders complex realities in 'objective' language, which carries normative weight and allows for clear metrics to aim towards (Hulme, 2020).

A 'way of seeing' not only influences how reality is represented but also can itself create reality ("singing the world into existence") (Lövbrand & Stripple, 2011; Paterson & Stripple, 2007). As such, to make a concept like climate change legible entails the production of new or revised knowledge, ideas, and objects that underpin the new legible system (Jasanoff,

[3] Humanity's response to climate change encompasses a variety of climate risk management interventions. This dissertation is particularly focused on climate mitigation; namely, "human intervention[s] to reduce emissions or enhance the sinks of greenhouse gases" (IPCC, 2023a, p. 126).

2018). Much of the process of making climate change legible has occurred at a global scale through institutions like the UNFCCC and the IPCC, with these bodies intentionally framing climate change as a 'global' risk (thus departing from earlier framings that emphasised local and regional impacts) (Allan, 2017; Fogel, 2005; Miller, 2004). Given that these processes are operating at a global level, a metric that could be globally legible and useful is required. The presentation of climate change as a scientific phenomenon led, in some way, to carbon becoming this global metric (Lövbrand & Stripple, 2011).

2.2. Carbon as the organising logic of mitigation

2.2.1. Scientific basis: why carbon becomes central

The framing of climate change has been called "intensely scientific" (Wynne, 2010, p. 291), and thus the way that the global science (which underpins the global mitigation governance regime) is represented is significant. The creation and representation of scientific knowledge is not neutral; rather, it is normative, with implications on what, for example, is implemented or researched (Allen et al., 2022; Jasanoff, 2010).

Lahn (2021) argues that the way in which climate change has been "enacted as a matter of public concern" has shifted over the last decade (p. 5). The UNFCCC's objective, formulated in the early 1990s, is the "stabilisation of greenhouse gases concentrations in the atmosphere" (UNFCCC, 1992, Article 2). This contrasts with the predominant conceptualisation of 'carbon budgets' (Lahn, 2021). This reflects a more nuanced understanding that it is cumulative net additions of carbon dioxide over time that determine greenhouse gas concentrations at any point in time and thus the associated induced warming (Allen et al., 2009; Matthews et al., 2009). The ramification is that halting humanity's anthropogenic emissions into the atmosphere will have the same effect on the concomitant rise in global temperatures. Entirely ceasing global emissions is challenging. A balance between emissions and removals thus becomes necessary to stabilise global temperatures at a certain level (Allen et al., 2022; Fankhauser et al., 2022).

These scientific understandings are directly used in the process of making climate legible. In its fifth assessment report, the IPCC "highlighted the cumulative amount of carbon emitted from human activities as the *relevant metric for climate policymaking* (emphasis added)" (IPCC, 2014; Lahn, 2021, p. 5). This is directly taken up in the Paris Agreement, in which the international community revised the objective of global mitigation; viz. to "aim to reach global peaking of greenhouse gas emissions as soon as possible… and to undertake rapid reductions thereafter… so as to achieve a *balance* between anthropogenic emissions by sources and removals by sinks of greenhouse gases in the second half of this century (emphasis added)" (Paris Agreement to the United Nations Framework Convention on Climate Change, 2015, Article 4). In other words, to achieve the Paris Agreement's temperature goals, net zero carbon by mid-century is necessary (IPCC, 2018). The emphasis thus becomes the balancing of flows of carbon.

2.2.2. Carbon: the global currency of mitigation

Carbon therefore becomes a kind of global currency for climate mitigation, rendering the 'wicked problem' legible by using the objective authority of its scientific status as a driver of climate change (Allen et al., 2009; Lövbrand & Stripple, 2011). Carbon also creates legibility through allowing processes of quantification, standardisation, and commensuration (Espeland & Stevens, 1998; Lazarus, 2008; Scott, 1998). These processes permit the comparison of otherwise disparate emissions sources – from power generation to deforestation – across time (Espeland & Stevens, 1998; Lövbrand & Stripple, 2011). This, in turn, allows target setting, tracking of progress, as well as substitution and exchange, such that emissions can be, for example, offset or traded through market-based mechanisms

(Gifford, 2020; Lang et al., 2024; MacKenzie, 2009). This has its benefits, allowing for global co-ordination and a clear way forward in the face of complex realities.

2.2.3. Governance implications: atomisation and scalability

The use of carbon as a metric impacts the way that mitigation governance is structured. Since carbon is fungible and scalable, governance and targets can be ‘atomized’ (Welton, 2022, p. 228). Consequently, from the UNFCCC to the Paris Agreement, there has been a sea change in governance that moves from the top-down approach to a bottom-up framing (Falkner, 2016). Under the UNFCCC, global mitigation was approached in a ‘regulatory’ manner, and thus enacted through negotiated, shared emission reduction targets (Hale, 2020). Paris has changed this narrative. The treaty requires each country to submit a nationally determined contribution (NDC), which functions as an individual, nation-specific target. It also expanded its realm of influence by increasing the role of sub-national actors and corporates (Hale, 2020). Many of these actors have also set net zero targets, which are framed in the same individualised way as the NDCs (Lang et al., 2024).

The cleanness of the new net zero framing has contributed to its rapid evolution to become the “organising paradigm of climate [mitigation]” (Welton, 2022, p. 3). An oft-quoted statistic puts 16% of global GDP under a national net zero target in mid-2019; four years later it had reached 91% (Hans et al., 2022). At the time of writing, a broad swath of the global economy – some 139 nations, 222 regions in the world’s highest-emitting countries, 333 of the most populous cities, and 1,280 of the Forbes 2000 companies – had set net zero targets (Lang et al., 2024).[4] Governance thus is no longer a global, coordinated effort as had been the case under the UNFCCC. Instead, the Paris governance regime is modular, in line with carbon’s fungibility. Carbon thus structures how mitigation is framed and implemented.

[4] The Forbes 2000 represents the largest 2000 publicly listed companies by revenue. At the time of writing, the share of global GDP covered by a net zero target had declined to 77%, largely owing to the Trump administration’s scrapping of the U.S. climate target (Pooler & Stott, 2025).

2.2.4. Abstraction as a result of carbon-centrism

To function as a global metric, carbon must be standardised, decontextualised, and comparable. The foundation of this proposition is that, since the atmosphere is global, a reduction of carbon in Johannesburg or Jakarta is functionally equivalent. Thus, using carbon as the sole metric allows for it to be irrelevant where or who or how the reduction is made. Accordingly, mitigation projects are abstracted from their "place, technology, history and [greenhouse] gas" in order to reduce them to their CO_2-equivalent flows (Lohmann, 2009, p. 80). This is bolstered by the fact that climate science and mitigation policy exist as a kind of global knowledge; that is, knowledge that "erases geographical and cultural difference and in which scale collapses to the global" (Fogel, 2005; Hulme, 2010, p. 559). It thus follows, Lohmann (2009) argues, that the wider impacts of a project become equivalent to another's. To follow this conclusion to its end is to reach the point of absurdism. The governance, ecological, and social impacts that arise from a mitigation activity cannot be functionally equivalent,[5] given the local contingencies that attach to them. Ignoring these wider backgrounds at the expense of carbon can have a number of ramifications, explored in the next section.

2.3. What carbon-centric mitigation obscures

While the governance scaffolding for net zero exists at a global level, implementation must ultimately occur at a local scale. The net zero project therefore requires the translation of a global target to the complexity of localised host communities and ecosystems, with competing or contradictory drivers, governance structures, incentives, and values (Hulme,

[5] As Barry Lopez so vividly writes, "Anyone determined to see so many of the world's disparate faces might easily succumb to the heresy of believing one place is finally not so different from another somewhere…But each place is itself only, and nowhere repeated… It is now my understanding that diversity is not, as I had once thought, a characteristic of life. It is, instead, a condition necessary for life. To eliminate diversity would be eliminating carbon and expecting life to go on. This, I believe, is why even a passing acquaintance with endangered languages or endangered species or endangered cultural traditions brings with it so much anxiety, so much sadness. We know in our tissues that the fewer the differences we encounter in our travels, the more widespread the kingdom of Death has become" (Lopez, 2013, pp. 17, 19).

2010). Welton suggests that our current framings of mitigation suffer from a “neutrality mirage”, in which the contestations around distribution considerations and how to achieve targets are hidden (Welton, 2022, p. 175). In other words, the translation of this global logic to local realities, combined with the use of a single metric (carbon) can render some of a wider set of values and services of the system as invisible (Espeland & Stevens, 1998; Hulme, 2010, 2019).

This dissertation is particularly interested in three interrelated dimensions of mitigation that can be obscured by purely carbon-centric approaches: mitigation’s interactions with other interventions and policy objectives; ecological constraints and trade-offs; and social and political consequences. Each is considered in turn.

First, a siloed approach to mitigation may obscure the fact that a specific mitigation effort is situated within a broader network of climate risk management interventions; that is, other measures that are aimed at decreasing risks accruing from climate change. This could be, for example, additional mitigation actions like those relating to emissions reductions or carbon dioxide removal (CDR), but also covers broader interventions like climate adaptation. If designed well, there is potential for synergies and co-benefits arising from these interactions (Girardello et al., 2019; Thornton & Comberti, 2017). However, the interactions may also involve trade-offs or have negative repercussions. For example, there is a well-established concern that a focus on CDR may create mitigation deterrence, in which CDR disincentivises emissions reductions (Grant et al., 2021; McLaren et al., 2023). Additionally, poorly designed mitigation may lead to maladaptation, with evidence of forestation projects in Cambodia leading to lower water levels in an already drought sensitive area (Work et al., 2019; Sharifi, 2020).

In addition, mitigation measures will interact with a broader set of policy objectives. Mitigation efforts, for example, can have either negative or positive impacts on biodiversity or development targets (Campagnolo & Davide, 2019; Jakob & Steckel, 2014; P. Smith et al., 2022). Accounting for these interactions is critical for ensuring that mitigation activities are not only effective in the short term but durable over time. Interventions that align with multiple policy objectives can build broader support (amongst the public and within institutions) and are less likely be to be contested or reversed (Meckling et al., 2015).

Second, global governance structures have rendered ecosystems in the language of 'carbon sinks' (or indeed potential sources) (Lövbrand & Stripple, 2011; Paris Agreement to the United Nations Framework Convention on Climate Change, 2015; UNFCCC, 1992). This is illustrative of a broader concern: carbon-centric mitigation can abstract the effort from the ecosystem in which it is embedded. This often manifests in a siloed focus on the maximisation of aboveground carbon (Aguirre-Gutiérrez et al., 2023). This is not say that co-benefits are entirely ignored: rather, it is often broadly assumed that increasing aboveground carbon stocks will always have collateral benefits for biodiversity and human development (Aguirre-Gutiérrez et al., 2023).

Yet the potential for these benefits is heavily dependent on context. Mitigation is embedded within, influences, and is dependent on ecological systems (Biggs et al., 2015). For example, the goods and services provided by ecosystems that are critical to many households' basic needs may be altered where mitigation is not sensitively designed (Edstedt & Carton, 2018; Fedele et al., 2021; Yang et al., 2013). A focus only on carbon can veil these dynamics, hiding ecological constraints, feedbacks, and trade-offs that impact the persistence of mitigation outcomes over time (P. Smith et al., 2019).

Third, the translation of the carbon imperative from global to local may have implications for host communities. The appeal of net zero lies in its tidy, customisable nature (Lin, 2022; Welton, 2022). As currently conceptualised, the target does not mandate a preordained emissions trajectory; rather, structuring the route to net zero has hitherto been left to the individual target setter, a flexibility enabled by the fungibility of carbon. In response, a number of authors have argued that this ambiguous quality hides critical social concerns and contestations around justice and equity (Armstrong & McLaren, 2022; Hulme, 2020; Khosla et al., 2023; Welton, 2022).

Welton contends that this false neutrality should be discarded in favour of a recognition that net zero is best characterised as a "social project" with distributional effects (Welton, 2018, 2022, p. 222). This has two implications. First, the social project has to operate at multiple scales, from the international policy realm to the hyper-local mitigation project on the ground. This entails translations between the scales (Carton, 2020), which may have ramifications for governance arrangements (Ostrom, 2010). Second, mitigation has implications for justice, including distribution of benefits, as well as impacts on tenure and access rights (Baker, 2021; Cavanagh & Benjaminsen, 2014; Sovacool, 2021). Thus, mitigation may worsen pre-existing, or create new, injustices (Muttitt & Kartha, 2020; Sovacool et al., 2019).

However, the current focus on carbon "largely ignores these social dimensions, which are critical for political legitimacy and durability" (Welton, 2022, p. 176). The consequences of not taking into account social aspects of a mitigation effort can be as serious as the de-legitimatisation of the net zero project. Indeed, "[m]aintaining public support through a three-decade transition to net zero simply cannot be achieved without the development and maintenance of a strong social contract" (National Academies of Sciences, Engineering, and Medicine, 2021, p. 1). In other words, these distributional effects are important for their own sake but also for the longevity of mitigation actions.

This section gave a high-level overview of what this framing could obscure (relevant to the conceptual nature of the first research paper); the following situates it within forestation as a case study (relevant to the two empirical studies).

2.4. Balancing the scales: carbon dioxide removal within the net zero paradigm

In this dissertation, two of the three research papers use land-based mitigation as a test case for the proposition that this type of mitigation impacts not only how much carbon is sequestered but also the social and ecological systems within which it is embedded (McGregor et al., 2019). The two papers specifically focus on global afforestation and reforestation, and urban nature-based solutions (including the urban canopy). Since both papers are broadly focused on carbon dioxide removal (rather than emissions reductions), this section considers the growth and potential implications of CDR.[6]

To achieve net zero will require rapid and deep emissions reductions across the global economy, but some sectors may prove challenging to completely decarbonise.[7] Thus, in order to achieve the balance required by the Paris temperature goals, modelled pathways show that any remaining emissions will have to be compensated by the removal of greenhouse gases from the atmosphere, either through removal technology or through the enhancement of the natural carbon cycle (IPCC, 2018). Net zero's focus on a balance between 'sources' and 'sinks' introduces a corresponding requirement: that any remaining emissions must be counterbalanced. Accordingly, the lead up to, and rise of, net zero has seen a concomitant rise in the perceived importance of CDR.[8]

CDR methods currently fall into two major camps, characterised by S. M. Smith et al. (2023) as *conventional CDR on land* (e.g. reforestation) and *novel CDR* (e.g. ocean alkalinisation).

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[6] In this dissertation, the IPCC's definition of CDR is instructive: "Anthropogenic activities removing carbon dioxide (CO_2) from the atmosphere and durably storing it in geological, terrestrial, or ocean reservoirs, or in products. It includes existing and potential anthropogenic enhancement of biological or geochemical CO_2 sinks and direct air carbon dioxide capture and storage (DACCS), but excludes natural CO_2 uptake not directly caused by human activities" (IPCC, 2022a, p. 121).
[7] Some current examples include agriculture, aviation, and some types of industry (IPCC, 2022c).
[8] This does not mean to suggest that net zero was the genesis of an interest in CDR. Indeed, as Carton et al (2020) note: "Carbon removal has been on the political agenda at least since the start of the UNFCCC negotiations in the beginning of the 1990s" (Carton et al., 2020, p. 5).

Conventional CDR projects are premised upon enhancing the natural carbon cycle, in which the ocean and land sequester carbon from the atmosphere.[9] In practice, the current CDR industry predominantly comprises conventional CDR, primarily afforestation, reforestation, and improved forest management (together, 'forestation') (S. Smith et al., 2024). This trend is evident in more than half of national net zero targets, as well as the composition of carbon markets (Climate Change Committee, 2022; Dooley et al., 2022). The literature, too, is replete with references to the climate mitigation potential of forestation (Canadell & Raupach, 2008; Chapman et al., 2020; Mildrexler et al., 2023; Nzabarinda et al., 2025). This can manifest as large-scale forestation in rural areas or small-scale efforts in urban areas (Edstedt & Carton, 2018; Sharma et al., 2024).

The popularity of forestation is explained by a number of drivers. It is partially as a result of its low cost relative to other CDR methods (Doelman et al., 2020); partially owing to aboveground carbon (as stored by trees) being easily understood and measured (Fleischman et al., 2020); and partially arising out of long-standing cultural attachments to trees (Pausas & Bond, 2019; Roman et al., 2021; Veldman et al., 2019). In addition, forestation, as well as other conventional CDR projects, are often framed as a 'triple win' with biodiversity and social co-benefits (Girardin et al., 2021; Muradian et al., 2013).

Critically, conventional CDR projects like forestation need land to exist: the total acreage required to meet pledged conventional CDR in national climate targets is equal to the current area of global croplands (Dooley et al., 2022). Land is intertwined with local social and ecological systems. Accordingly, there are a number of competing priorities for how we use our land, complicated by its boundedness (Popp, 2022). Valuing land for its carbon contribution may ignore other activities or values associated with the land (Hulme, 2010,

[9] For clarity, according to S. M. Smith et al. (2023), conventional CDR includes "afforestation/reforestation; soil carbon in croplands and grasslands; peatland and wetland restoration; agroforestry; improved forest management; and durable Harvested Wood Products" (p. 17).

2019). Not only is this an issue for decisions concerning land-use but also for the potential impacts of CDR on the host communities and ecosystems.

There are a myriad of ecological constraints on where forests may naturally occur, including permafrost, salinity, or length of growing season (Luedeling et al., 2019). Non-forest ecosystems include “deserts too dry for forests, tundra too cold for forests, and soils that are flooded or saline and hostile to tree growth” (Bond, 2019). Even where these constraints do not exist, disturbance regimes may structure the ecosystem towards being more open, such that a grassland, savanna, or open woodland may exist as an alternative stable state to a forest in that area (Staver et al., 2011).

To wit, naturally open ecosystems can exist in an uncertain climate space, which are “climate envelope[s] where vegetation is not at equilibrium with climate and either grassy or forest ecosystems can occur” (Stevens et al., 2022, p. 262). Accordingly, while forests could theoretically be planted in these areas, natural disturbances (e.g., fires or large herbivores) have maintained the ecosystem as largely comprising light-loving fauna and flora. This fact of evolutionary history and ecosystem function, however, is challenged by deeply ingrained narratives (often stemming from colonial interactions with the land) that characterise forests as the ‘natural’ state of ecosystems in areas where they might not be (owing to natural disturbances), while open ecosystems are often seen as deforested ‘wastelands’ (Pausas & Bond, 2019). The strong narrative forces, as well as the suitable climate, mean that researchers and practitioners may misidentify these areas as suitable for afforestation, even where this may be ecologically inappropriate (J. F. Bastin et al., 2019; CarbonParks, 2023; Mano, 2022; TotalEnergies, 2023; Zomer et al., 2008).

This can have serious consequences for the very metric that is being prioritised (carbon). Forestation in unsuitable ecosystems may lead to the loss of existing carbon stocks, including soil carbon, which would undermine the permanence of the sequestered carbon

(Stevens & Bond, 2024; Zhou et al., 2022). This raises concerns about the durability of these types of CDR projects. Given the reliance many net zero targets on carbon removals (Net Zero Tracker, 2023; S. Smith et al., 2024), the integrity of net zero targets is contingent on the ecological appropriateness of forestation efforts.

The durability of a forestation measure also depends on public support. There are a number of ways in which mitigation in the form of forestation may impact a social system (Newell et al., 2021). For example, forestation projects may have negative ramifications for local communities, such as the movement of residents off their historic land. Dispossession is already occurring in practice, with risks that this trend may continue into the future (Cavanagh & Benjaminsen, 2014). Indeed, large-scale forestation maps, which are used for the identification of areas for forestation projects, have been critiqued for increasing the risk of displacing marginalised people (Fleischman et al., 2022). In many places where forestation is popular with project developers, this has echoes of histories of external interventions, such as colonial or 'development' interactions (Leach & Scoones, 2015). Carbon-centrism contributes directly to this since often the price of carbon can be more valuable than current uses of the land (Fairhead et al., 2012).

This same pricing-out issue may also delineate access to ecosystems that were previously utilised by residents for ecosystem goods and services (Veldman, Overbeck, et al., 2015b). Alternatively, the introduction of forestation may impact the provision of these services. A study of a forestation site in Uganda, for example, found evidence that ecosystem services relied on by the community – provision of fuelwood, grazing, and water – were degraded by the project, leading to an increase in poverty in the area (Edstedt & Carton, 2018). This can affect public support for, and the associated durability of, the project.

Thus, these ecological and social concerns suggest that a purely carbon-centric framing does not simply fail to account for the wider system but may actively incentivise mitigation interventions that are misaligned with the context in which they are embedded.

2.5. Integrating mitigation and the wider context

A mechanism for avoiding the limitations of carbon-centric mitigation is the development of integrated approaches that situate mitigation within the wider socio-ecological context. Recent research has begun to situate mitigation within its context, or integrate with it individual drivers, but this work has been uneven.

For example, integrated assessment models (IAMs) were developed in the 1990s; while they do integrate a wider of concerns into climate mitigation, these still rely on a central metric of carbon (Pindyck, 2013; Rogelj et al., 2018; Welton, 2022). This means that the models typically only consider the impacts of the wider set of drivers on mitigation policy, without considering the impact of mitigation on those drivers (Riahi et al., 2017). Accordingly, while IAMs incorporate multiple drivers, they do not fully capture the bidirectional interactions between mitigation and the wider context.

Some parts of the literature more clearly recognise this. For example, the socio-ecological systems (SES) literature recognises the intertwined nature of ecological and social systems (Ostrom, 2009). Here the research body has developed theory on transformation of SES and the applicability of the framework to climate adaptation (Chaffin et al., 2016; Hossain et al., 2024; Moore et al., 2014); but focuses on mitigation are still lacking. At the same time, some authors have suggested, in the context of ecosystem services, that environmental governance should pay attention to social feedbacks (Dawson et al., 2017), while others have introduced the concept of 'sociocarbon cycles' to move beyond a focus on carbon, presenting forest governance as a socio-natural process (McGregor et al., 2019).

Nature-based solutions for mitigation offers the clearest example of the work done to situate mitigation within ecological and social systems. This body of literature integrates natural climate solutions with biodiversity and human well-being, recognising that cooperation and coordinated action are critical for this integration (Seddon et al., 2020). However, while nature-based solutions integrate mitigation with ecological and social objectives, there are gaps in how this concept applies in specific socio-ecological contexts. Indeed, the concept has been critiqued for being too vague for operationalisation (Olivadese, 2025), particularly in urban contexts. While tools to concretize urban NbS in specific settings are being developed (Raymond et al., 2017), there is still work to be done to make the concept applicable to specific contexts. For example, as is discussed further in the introduction to **chapter 5**, there is little clarity in the literature on what the ecological baseline for urban NbS should be where the novel ecosystem that has co-evolved with the city differs from the surrounding native ecosystem.

When considered together, these strands of the literature suggest that there is an emerging recognition of the need for integration but that there still remains some fragmentation between mitigation research and that focused on social and ecological systems. Integration between these is often stymied by institutional siloes as a result of separate knowledge development (Ostrom, 2009). The way that knowledge is presented has implications for the risks that are perceived, and thus managed, by policy makers (Beck & Mahony, 2018; Carton et al., 2020). This fragmentation reinforces carbon-centric approaches by privileging simplified metrics over the broader system. This may limit the extent to which mitigation can be designed as part of a wider socio-ecological system, affecting durability.

2.6. Literature gap summary

Current approaches to mitigation focus on carbon as a key metric of success. However, such a focus may miss a wider set of interactions with the socio-ecological context within which the relevant mitigation effort is situated. These interactions can have impacts on other policy issues that we care about (e.g. human well-being; ecosystem health) but in addition may threaten the durability of the mitigation intervention itself. A potential solution to this issue is the use of integrated approaches, which have emerged in the literature. However, the application to mitigation has been patchy or has lacked the tools for implementation at a project level. The following chapters respond to these gaps in the literature in the following ways: **chapter 3** situates mitigation within a wider system of climate risk management interventions to capture the interactions between interventions. **Chapter 4** is predominantly focused on quantifying the extent to which forestation projects have not been situated in an appropriate ecological system. Where these projects are misplaced, this may have negative impacts on the ecosystem, as well as the forestation itself, which will be vulnerable to the constraints that may maintain the ecosystem as non-forested. **Chapter 5** situates a mooted mitigation effort (forestation) in an urban landscape in Johannesburg, South Africa. This process of situating requires a comparison between the native ecosystem (grasslands) and the novel ecosystem (urban forest), as well as the social implications (distribution of benefits across race and socio-economic status) between increasing either. This uses the nature-based solutions paradigm but critically extends it by showing how the paradigm can be operationalised in an urban setting where the native vegetation contrasts with the proposed natural intervention.

3. A Climate Risk Management Typology: Integrating Approaches to Reduce Risk

Make the circle bigger – JR (ft. HHR)

The first research paper in the thesis presents a new typology for climate risk management, illustrating a conceptual framework for an integrated approach. We present a toolkit for using climate risk management interventions to reduce risk, presenting it as a tool that can be customized for varying risks, localities and time scales. We further present a set of six portfolios as an example of the typology in practice. This typology serves to situate individual mitigation efforts (such as CDR) within a broader governance system of other mooted climate risk interventions (such as emissions reductions, adaptation, or solar radiation management), thus capturing interactions, trade-offs, and synergies.

The article was published as part of an 'article forum' in which researchers were invited to respond to our paper, after which we could provide an additional response. Our response article is included as Appendix II.

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A Climate Risk Management Typology: Integrating Approaches to Reduce Risk

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Overshoot of the Paris Agreement's 1.5ºC temperature goal is increasingly likely, resulting in climate risk management becoming more complex. This is aggravated by an often siloed and fragmentary risk management landscape, making the risks and benefits of potential interactions between interventions less legible to practitioners, policymakers, and researchers. Approaches that integrate thinking across all climate risk management interventions are emerging as an increasing priority. We present a qualitative climate risk management typology that aids in exploring the interactions and roles of climate interventions across a variety of climate risk scenarios and contexts. Use of the typology is demonstrated by presenting six portfolios of interventions, which illustrate the utility of different interventions to ultimately reduce loss and damage. This approach builds on existing tools in the literature by recasting them explicitly in terms of risk and extending their range over a variety of qualitatively distinct climate scenarios.

Key words: Overshoot, risk management, mitigation, policy, framework

3.1. Introduction

Achieving the 1.5ºC temperature goal enshrined in the Paris Agreement requires rapid and deep decarbonisation (IPCC, 2023c). The remaining carbon budget for limiting warming to 1.5ºC is rapidly shrinking, with the level of human-induced global warming already at 1.31ºC above pre-industrial levels in 2023 and current rates of warming (as at the time of writing) placing the world on a path to reaching 1.5ºC by the mid-2030s (Forster et al., 2024). Even before the 1.5ºC target is met, natural variability means that individual years are increasingly likely to temporarily exceed 1.5ºC (Hermanson et al., 2022), with 2024 already reaching an average of 1.55ºC ± 0.13 °C (World Meteorological Organization, 2025). Therefore, risks resulting from short-term warming of 1.5ºC will soon start to emerge, often in more vulnerable and exposed socio-ecological systems, before chronic impacts from sustained warming above 1.5ºC are realised over the long-term (IPCC, 2021, 2022b).

Thus, while not inevitable, it seems increasingly likely that limiting warming to 1.5ºC will entail an overshoot, in which the peak global temperature initially exceeds 1.5ºC before returning to, or below, this target (Climate Overshoot Commission, 2023; IPCC, 2023c; United Nations Environment Programme, 2024). The greater the delay in decarbonising the global economy, the greater the scale and speed required from various climate interventions to ultimately limit long-term warming to 1.5ºC (IPCC, 2023c). As a result, interventions that are discussed as having especial utility in overshoot scenarios – including greenhouse-gas removal, solar radiation modification, drastic emissions reductions, or extreme adaptation – are attracting increasing interest (Browne, 2022; Fialka, 2020; Fuss et al., 2018; Long & Shepherd, 2014; MacMartin et al., 2018; McLaren & Corry, 2021; Morrison et al., 2022; Reuters, 2022; Rogelj et al., 2018; S. Smith et al., 2024; United Nations Environment Programme, 2023; Warszawski et al., 2021).

These developments are emerging in an existing climate risk management system, which historically has tended towards a siloed or even fragmented approach to reducing risk. Evidence of this is clear in international climate structures: recall the separate working groups of the IPCC or the differentiated negotiation streams of the UNFCCC (Stocker & Plattner, 2014; Wamsler et al., 2020). Interactions between potential climate interventions can thus seem to be a zero-sum game; for example, greenhouse gas removals and drastic emissions reductions are often positioned in opposition to each other (Brad & Schneider, 2023; Grant et al., 2021; McLaren et al., 2023).

While such trade-offs between interventions can exist, particularly in the context of resource constraints, interventions may also interact in ways that are synergistic (Warren, 2011). Further, since interventions are systemically connected, approaches that do not holistically consider interventions and their interactions may fail to capitalise on potential synergies or, indeed, create more risk as a result of unexpected interactions (Grafakos et al., 2019). We argue that the increasing urgency and scale of the requisite global climate response means that interactions between interventions are increasingly non-trivial and must therefore be carefully considered.

There has been a growing recognition that this exercise might be aided by systemic approaches that synthesise and integrate thinking across these interventions (see, as one example, the IPCC's Synthesis Report) (Grafakos et al., 2019; IPCC, 2023c; Wamsler et al., 2020) and we aim in this paper to contribute towards this nascent movement. Accordingly, we present a qualitative tool to construct risk management *scenarios* that visualise the magnitude of, and potential responses to, a particular risk. To build a scenario requires considering six '*high-level factors*' that affect the management of the risk. The former three high-level factors set out the magnitude of the risk in question; the latter three are *interventions*, which are actions that aim to decrease the risk. Each high-level factor has a set of archetypal '*instances*', which are qualitatively distinct ways in which the high-level factor might manifest. '*Portfolios*' are combinations of interventions that respond to and exist within a particular risk scenario. Presenting all interventions within one diagram, and specifically in terms of risk, is intended to act as a stimulus for risk management discussions, as users are required to assess the various interventions, impacts, dynamics, and interactions, within an intervention portfolio for a given risk management context.

We provide an illustrative example of six scenarios constructed using this typology, and a method for creating sets of scenarios to support other risk management questions. This climate risk management typology builds on the 'napkin diagram' proposed by Long & Shepherd (2014), extending it to accommodate a range of futures and interventions. Crucially, the typology proposed here recasts warming scenarios in terms of *risk*, instead of *temperature*. This shift front-lines important discussions on how to characterise the risk (Reisinger et al., 2020), the complex components involved in assessing it, and the values and uncertainties that can often be hidden while focusing on temperature outcomes.

We present example diagrams to illustrate one particular way of thinking holistically and systemically but emphasise that they are a small subset of the much larger space of possible scenarios and portfolios laid out here and made available by the full typology. By combining 'instances' from each 'high-level factor' when analysing risk management futures, scenarios can be crafted for other values, scales, and subjects as required. In this paper we focus on climate change, but the typology could be equally well deployed for other risk management challenges, such as those related to biodiversity or public health.

A qualitative tool can be a useful addition, both to help frame quantitative work, and create an agenda in research that helps eliminate blind spots. A tool like the climate portfolios aids in highlighting possibilities and risks but is not designed to make final decisions; in that way, it is a map rather than a navigator (Edenhofer & Minx, 2014). Indeed, the typology can be particularly useful in the setting of collaborative workshopping exercises, particularly those comprising a broad set of stakeholders. Using the typology to frame discussions about risk might aid stakeholders in sharing softer knowledge and coming to common understanding of the interactions between each component in a diagram (Mechler et al., 2019).

Linked to this, no tool is value-free (the role of the mapmaker is not apolitical), and we recognise that the scenarios touch upon a number of contested subjects (Cox et al., 2020; Kopp et al., 2025; McLaren & Corry, 2021; Waller et al., 2020). However, in requiring consideration and explicit representation of the risks and uncertainties involved with a particular choice of intervention (rather than temperature impact alone) when using the typology to build scenarios, we hope that the tool may be useful in underlining the calls for a global moratorium on the deployment of methods that as yet are insufficiently understood, such as solar radiation modification (Biermann et al., 2022), as well as the need for a more integrated global governance system to coordinate workstreams.

3.2. The Climate Risk Management Typology

The climate risk management typology introduced in this article builds on Long & Shepherd's "napkin diagram" (Long & Shepherd, 2014), see **Figure 3** below, which was created to illustrate one potential role of various interventions (including emissions reductions, greenhouse gas removal, and solar radiation modification) in limiting the rise of global mean temperatures, and the roles of adaptation in reducing the remaining impacts on loss and damage. The napkin diagram is positioned at a global level and considers a timescale over two hundred years (between 2000 and 2200). Its clarity in conveying its message has meant that it has been adopted in the climate community and literature (Belaia, Borth, et al., 2021; Belaia, Moreno-Cruz, et al., 2021; J. L. Reynolds, 2019).

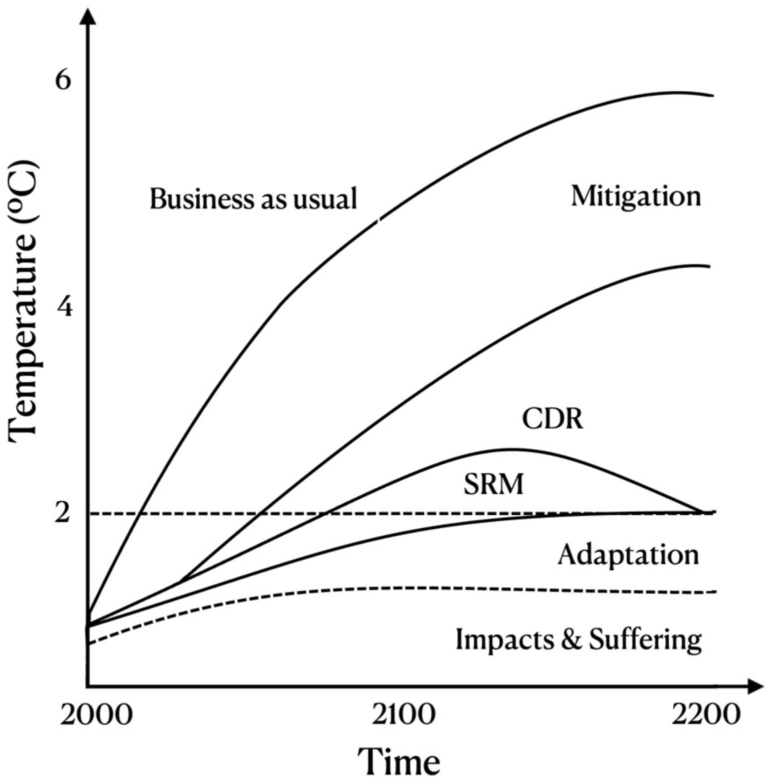


Figure 3: The Napkin Diagram. *The diagram was created by Long and Shepherd to illustrate that climate change should be responded to via a portfolio of approaches, incorporating emissions mitigation, carbon dioxide removal (CDR), solar radiation modification (SRM), adaptation, impacts and suffering. Based on J. G. Shepherd (n.d.), with permission.*

The typology introduced here extends the initial “napkin diagram” in three ways. First, it explicitly considers both overshoot and non-overshoot pathways, which increases the relevance of the napkin diagram to the current state of the climate system and action (IPCC, 2023c). Second, the typology extends the diagram by explicitly incorporating a range of qualitatively distinct trajectories for mitigation and adaptation responses over time (as laid out in **Table 1**), instead of the single scenario in the napkin diagram (which depicts overshooting then net negative emissions, with temporary solar radiation modification deployment to flatten the underlying overshoot, and adaptation increasing over time before stabilising); the tool introduced here therefore provides greater clarity in the role of all available options and how they might interact. Third, we modify the napkin diagram by recasting the primary variable as *risk* instead of *temperature*, which has a number of important benefits for integrated risk management approaches.

First, this risk framing accounts for uncertainty in the temperature-risk coupling (Seneviratne et al., 2018) and also increases the relevance for decision makers for whom risk outcomes are ultimately what matter most. Second, this focus on *risk* enables consideration of risks relating to a variety of systems and geographical scales (by choosing a locally defined risk indicator, for example), and the co-dependence between local impact and *global* and *local* actions to be concisely expressed. Third, this risk framing adds utility by integrating the outputs of the three working groups of the IPCC; the typology represents at a high level the risks generated by changes in climate-related hazards (WGI), the coupling between climate hazards and risk (WGII), and the relationships between potential risk reductions from mitigation (WGI and WGIII), adaptation (WGII), and solar radiation modification, and the consequent risk of loss and damage (WGII).

Multiple types of risk emerge from climate change, and their exploration is supported by this typology. We take as our starting point the IPCC’s understanding of risk: the “potential for adverse consequences for human or ecological systems", determined by the interactions between hazards, vulnerability and exposure (IPCC, 2023c). We define physical climate risk as the emergent risks due to the state of the physical climate system after all mitigation activities that physically alter its composition or state have been factored in, but before the resulting risks to society have been reduced through adaptation. The specific physical climate risks that would be incorporated are dependent on the choice of risk variable in a scenario. We define societal climate risks as the remaining risks faced by society after adaptation measures have reduced the exposure and vulnerability to hazards from the physical climate, a definition therefore synonymous with the risk of loss and damage (Roberts & Pelling, 2018). An intervention is defined here as an activity explicitly taken to minimise physical or societal climate risk and is therefore additional to activities that might have occurred anyway without any risk-limiting intention but may reduce risk as a side effect. For example, carbon dioxide removed from the atmosphere through active reforestation efforts would be included, whereas removals resulting from carbon fertilisation (i.e. the increased carbon uptake of plants that comes along with increased atmospheric CO_2 concentrations) would not be (Nolan et al., 2024).

3.3. High-level factors: the building blocks of a risk management scenario

A selection of six climate risk management portfolios is presented in the next section; these were constructed using the climate risk management typology that is detailed in this section. The portfolios were chosen to illustrate, at a high level, the potential role of various interventions in addressing risks resulting from climate change. However, the utility of the typology is that it allows for the construction of arbitrary sets of portfolios that support risk management for a much wider range of scenarios. Here the typology is focused on climate change but could be directed at other risk management contexts.

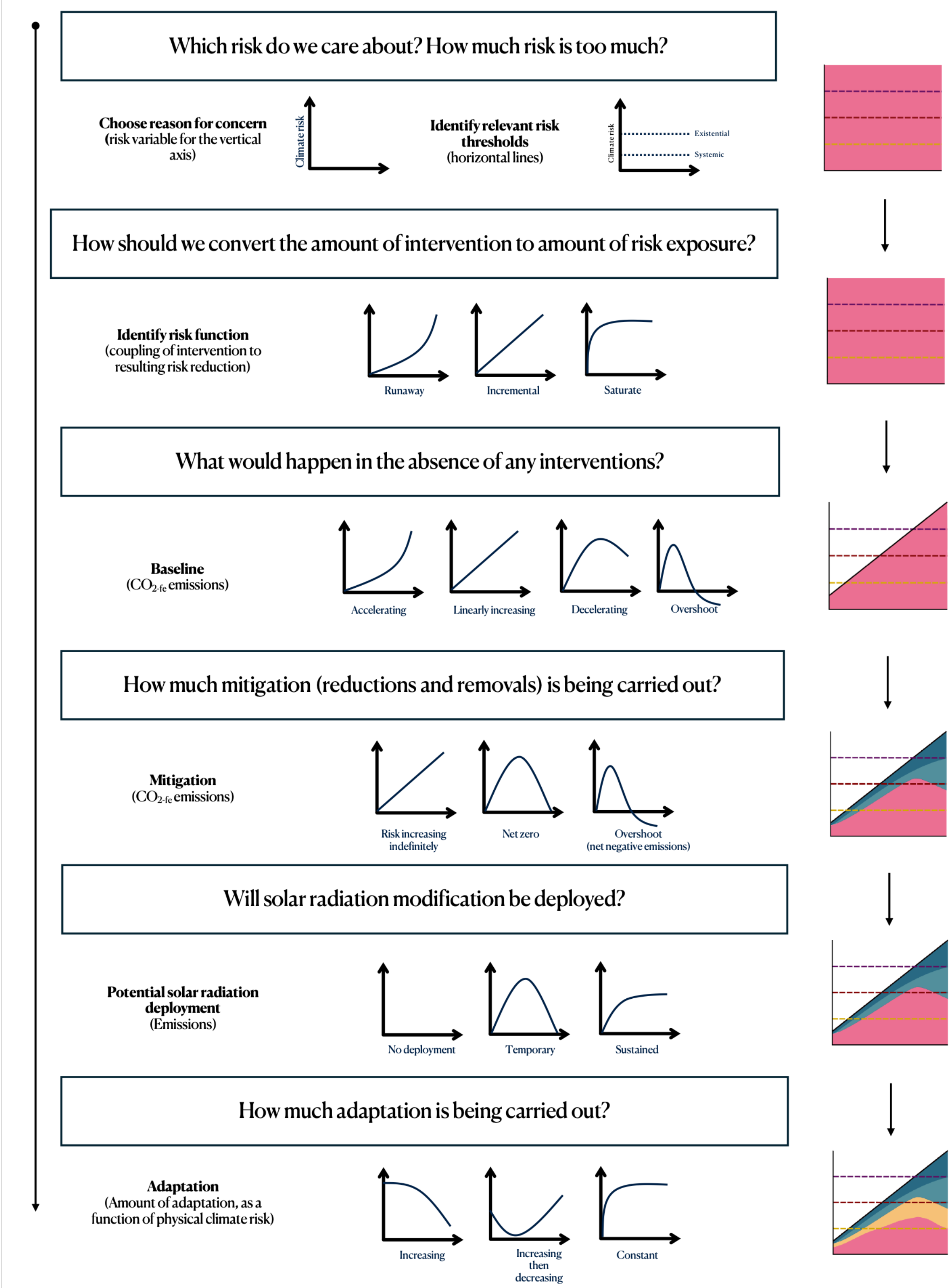


Figure 4: Constructing a climate risk management scenario. *Here we set out the process for creating a climate risk management scenario through six steps. Each step corresponds to a high-level factor (explained further in the main text and in* ***Table 1****). Choices must be made at each step about which 'instance' of the high-level factor will apply (instances shown here as graphs). To illustrate how this coheres to a scenario, one example representation of a climate risk scenario (****Figure 5****'s 'Plan B' scenario) is included on the right of each step; further climate scenarios are included in the main text. To construct a climate risk management scenario, a specific risk is chosen for the vertical axis (risk represented here as the choice of axis), and any useful thresholds for this risk can be depicted with horizontal lines (represented here as three lines in purple, red and orange) (step 1). A risk function must be selected to convert the adoption or absence of interventions into quantity of risk increase or decrease (step 2). Following this, a baseline trajectory must be selected to use as a benchmark against which interventions can be measured (here the linearly increasing instance of the baseline is chosen*

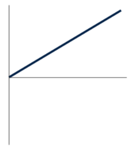
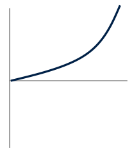
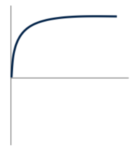
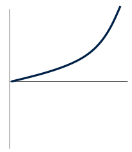
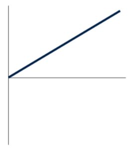
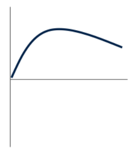
and is represented in the graph by the black line. Note how this effects the magnitude of the risk, effectively setting up a risk 'envelope' that is relevant for the scenario) (step 3). From this, interventions are inserted to decrease the risk within that envelope (steps 4-6). First, mitigation interventions are inserted (blue wedges), followed by solar radiation management where relevant (teal, here not inserted); and finally, adaptation inventions (here in yellow).

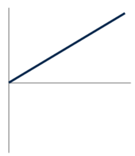
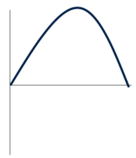
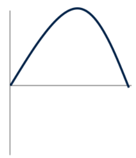
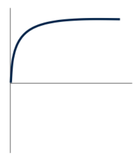
In this section, we present the typology of six high-level factors that can be used to build any generalised climate scenario as follows and as is presented in **Figure 4**. First, the type of risk is chosen for the vertical axis, and potential threshold levels of risk are identified and plotted as horizontal lines. Further, since the figure is cast in terms of risk, a risk function is required to convert the various amounts of a given intervention into risk outcomes for the given risk type. Next, the baseline trajectory is selected in order to give a baseline of what the outcome might have been in the absence of any interventions. The mitigation wedge is then included, based on trajectories of emissions, reductions, and removals. The deployment (if at all) of solar radiation modification is inserted between the mitigation and adaptation wedges. Lastly, the adaptation wedge is inserted, dependent on perceived ability and capacity of adaptation measures to decrease the level of vulnerability and exposure to physical risk over time. The remainder of the graph under the adaptation wedge refers to the risk of unavoidable loss and damage accruing from the risk type that could not be avoided or adapted to.

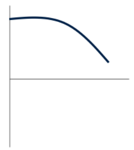
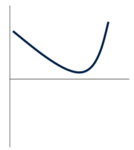
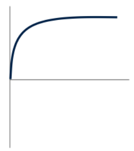
For each of the high-level factors, a stylised set of qualitatively distinct 'instances' are identified, which together span the set of baselines, interventions, climate responses, and risk outcomes that we perceive as likely to be strategically relevant. The high-level factors and varying instances are set out in **Table 1** and explored further below.

Specific archetypal instances of the high-level factors can be combined to create a climate risk management scenario, which illustrates the portfolio of measures that are necessary to ultimately decrease the risk of loss and damage. From the combination of instances in **Table 1**, there are approximately one thousand potential qualitatively distinct combinations, though not all are possible, likely, or deemed to be relevant.

Table 1*: High-level factors and their instances in the climate risk management typology. This table sets out the climate risk management typology that aids in exploring the interactions and roles of climate interventions across a variety of climate risk scenarios and contexts. The approach to risk management is structured through the use of 'climate risk management scenarios', which contain six 'high-level factors' that might be useful in understanding the magnitude of, and responding to, the relevant risks. A short description of each high-level factor is set out in this table. Each high-level factor has a set of 'instances' attached to it. The instances span the set of baselines, interventions, climate responses, and risk outcomes that we perceive as likely to be strategically relevant in constructing risk management scenarios. The first three high-level factors ('Reason for Concern and Risk Thresholds'; 'Risk Function'; 'Baseline') show the type and magnitude of the risk. The final three high-level factors ('Mitigation'; 'Potential Solar Radiation Management Deployment'; 'Adaptation') are 'interventions'; that is, mechanisms for responding to the risk in question. A combination of interventions creates a 'climate risk management portfolio'.*

Step	High-level factor	Description	Instances
1A	Reason for concern	**Choose reason for concern as the risk variable for the vertical axis.** Risk is chosen over other variables (e.g. temperature) in recognition that much of our response to climate is structured around decreasing risk.	The risk chosen can be customized to suit a variety of spatial and temporal scales. Risks could be global (global risk of extreme weather events occurring) or local (a city's risk of flooding due to sea-level rise over time).
1B	Risk thresholds	**Identify any relevant threshold levels as horizontal lines on vertical axis.** This recognises that there may be certain thresholds that may change the magnitude or pace of the risk and are thus relevant to how interventions might be structured.	Examples of threshold setting could include the planetary boundaries framework or estimated thresholds for tipping points that may exist in physical or socio-economic systems.
2	Risk function	**The risk function recognises that a conversion from the quantity of a given *intervention* to the quantity of *risk* may be necessary as the relationship is not always linear.** To calculate the risk-reduction wedge shapes for each intervention, there are two options: i. work forwards from a proposed amount of an intervention, using the risk function to calculate the resulting level of risk reduction over time to plot in the scenario, or ii. work backwards from a risk reduction wedge shape required in a scenario, using the inverted risk function to calculate the amount of the intervention required to give those risk reductions.	*We present three instances of the risk function:*  **Incremental**. Risk is monotonically coupled to the quantity of the intervention.  **Runaway**. Risk may initially increase 'incrementally' (as in the first instance) but past a certain point it continues to increase, regardless of interventions.  **Saturate**. Risk increases 'incrementally' but, after a certain point, ceases to increase, or may even decrease, as the risk may saturate beyond a certain point.
3	Baseline	**The risk reductions resulting from the various interventions must be measured against a baseline, so each climate risk management scenario requires the selection of a baseline trajectory.** In a climate risk management scenario, the baseline represents CO_2-fe emissions.	*We present four instances of potential baseline trajectories:*  **Accelerating**  **Linearly increasing**  **Decelerating**  **Overshoot**

Step	High-level factor	Description	Instances
4	Mitigation	**This high-level factor is focused on interventions that aim at mitigating against the driver of the risk. While the driver of the risk causes risk, eliminating the driver may not entirely decrease risk overall as residual risks may still occur.** In a climate risk management portfolio, for example, mitigation represents the pathway that anthropogenic forcing takes over time as a result of the amount of CO_2-fe emissions in the atmosphere. This includes both emissions reductions and greenhouse gas removals.	*We present three instances of mitigation intervention trajectories:*  **Risk increasing indefinitely.** Mitigation interventions fall short of halting the cause of the risk, which results in this cause of the risk following an indefinite upwards trajectory.  **Net zero**. Mitigation interventions cause the driver of the risk to decrease to, and then remain at, net zero.  **Overshoot (net negative emissions)**. Here mitigation interventions cause the driver of the risk to decrease to net zero and continue into net negative territory.
5	Potential solar radiation management	**This high-level factor is specifically relevant to climate risk scenarios and recognises that some pockets of the climate community have advocated for its inclusion in climate policy.** Inclusion of it in this typology is not an endorsement of the intervention. Rather, it aims to be clear on the risks associated with such an intervention being included in climate risk management portfolios.	*We present three instances of potential solar radiation deployment:*  **None.**  **Temporary**. A short-term period of solar radiation modification deployment is used to temporarily reduce risk in a given climate scenario.  **Sustained**. Continued deployment is used to keep temperatures, and therefore temperature-related risks, below certain thresholds on an ongoing basis.

Step	High-level factor	Description	Instances
6	Adaptation	**This intervention recognises that risks may have already arisen, despite mitigation attempts, and thus adaptation to actual or expected risks may be necessary to moderate the impact.** The risk reductions from adaptation interventions incorporate changes to learning over time as a result of improvements in both skill and capacity. The shape of the graphs corresponds to the quantity of risk that the adaptation intervention is causing to decrease (and not the quantity of the adaptation measure itself).	*We present three instances of adaptation over time:*  **Increasing.** The community in question learns over time and is able to increase its adaptation levels, even in a world where the driver of the risk remains constant or increases. The overall risk level therefore decreases.  **Increasing then decreasing**. The community is initially able to adapt to low levels of risk, but over time is able to adapt less and less well, perhaps because exposure increases (e.g., due to a growing population), or owing to the existence of limits to adaptation as the driver of risk increases.  **Constant.** The risk reductions from adaptation interventions remain constant over time. Note: this constancy need not correspond to a constant amount of adaptation activities.

The climate risk management scenarios vary depending on the qualitative structure of interventions required for limiting risk to a certain level, rather than quantitative differences in the amount of activity relating to each intervention within scenarios that use the same selected set of instances from the high-level factors. Potential interactions between different types of intervention are accounted for by the interventions being combined into a portfolio; this allows potential concerns over resource constraints to be considered, for example.

While the typology can be applied to globally averaged risks and related metrics, it can also be applied in a way that explicitly addresses local contexts and geographical nuance by carefully defining the risk considered on the vertical axis. For example, the drought risk to a city could be used as the risk variable on the vertical risk, with incremental, systemic, and existential thresholds relevant to the particular urban area. Mitigation and solar radiation modification deployment could be global activities in the scenario, while the locally specific realisation of physical climate risk, adaptation responses to the given risk, and resulting remaining loss and damage would apply specifically to the city's locality. This helpfully expresses co-dependence between global and local actions and unifies them in one visualisation for risk managers to demonstrate how global activities would link with locally realised impacts.

3.3.1. Risk type and thresholds

When building out a scenario using the typology, the preliminary step is to identify the extent or type of risk considered. Conceptually this underpins the entire exercise; visually, it is reflected as the variable on the vertical axis. Any risk could be chosen: for example, the burning embers employed by the IPCC to visualise key Reasons for Concern (as presented in IPCC AR6 WG2 SPM Figure SPM.3) (IPCC, 2022b; Marbaix et al., 2025; Zommers et al., 2020). Note that beyond global concerns (such as the global risk of extreme weather events occurring, risk to the stability of large-scale components of the Earth system, or global biodiversity loss), more localised risks may also be used (such as a city's risk of flooding due to sea-level rise over time, risks of local crop loss due to droughts, or the climate-induced risk of local increase of vector-borne diseases), which broadens the utility of the typology across a range of scales and risk-related problems.

Given a particular risk variable, it can be useful to set stylised risk levels as a visual mechanism for incorporating into risk management any concerns relating to the degradations or thresholds of risk. These correspond to certain levels of risk on the vertical axis and can therefore be depicted as horizontal lines on the figure (e.g., **Figure 5**). For our purposes, we have categorised risk thresholds into three degradations: incremental; systemic; and existential. However, the boundaries between each risk threshold are not certain or set, so care should be taken while choosing and communicating them. Examples of threshold setting could include the planetary boundaries framework or estimated thresholds for tipping points that may exist in physical or socio-economic systems (Armstrong McKay et al., 2022; Malhi et al., 2009; Steffen et al., 2015).

3.3.2. Risk function: coupling between interventions and risk reductions

After identifying the type and extent of the risk, the second step is to account for the coupling between interventions and risk reductions, through the risk function. This coupling incorporates complexity in the relationship between a given intervention and the resulting risk reduction, which requires careful consideration of the uncertainties, dynamics, and potential nonlinearities in the system connected with the specific risk under consideration. This coupling acts as a risk function that converts the *amount* of an intervention carried out to the *risk reductions* resulting from those activities. As such, to work out the risk-reduction wedge shapes for each intervention, you can either: (i) work forwards from a proposed amount of an intervention, using the risk function to calculate the resulting level of risk reduction over time to plot in the scenario, or (ii) work backwards from a risk reduction wedge shape required in a scenario, using the inverted risk function to calculate the amount of the intervention required to give those risk reductions. This can be done through the use of mathematical formulas or, at a higher-level, as a qualitatively estimated change in shape of the relevant wedge.

For activities in the portfolio (including the baseline activities, mitigation interventions, and solar radiation modification interventions) where risk and risk reductions are mediated through the physical climate system via the emission, reduction, and removal, of various climate pollutants, the coupling includes components relating to both (i) how the specific risk is coupled to the state of the climate (the "risk response"), and (ii) how the state of the climate is, in turn, coupled with emissions and other anthropogenic forcings (the "physical climate response").

While the relationship between cumulative emissions and global temperature has been roughly linear to date (MacDougall, 2016; MacDougall & Friedlingstein, 2015; Matthews et al., 2009), explicitly including the "physical climate response" enables scenarios to be built that incorporate concerns that non-linearities that may exist in the relationship between forcing and the state of the climate (Huang et al., 2022; Nicholls et al., 2020). For many risks types, the "risk response" that connects the state of the climate to impacts can be challenging to identify and quantify to a high level of precision and confidence due to, for example, nonlinearities and regional variations (IPCC, 2022b).

The risk reductions resulting from adaptation interventions are more direct (but not necessarily linear), as the nature of these activities and their impacts tend to be more local, specific, and not mediated via the global climate system (IPCC, 2022b; T. G. Shepherd & Sobel, 2020).

Three stylised examples of coupling are included in the typology in **Table 1**. In the first ('incremental'), risk is roughly monotonically coupled to emissions; in this case risk increases as a function of anthropogenic forcing but stops increasing approximately when anthropogenic forcing stops. In the second ('runaway'), risk may initially increase 'incrementally' but past a certain point it continues to increase even after anthropogenic forcing ceases. An example of this would be if the chosen risk type featured tipping behaviours (like AMOC) (Armstrong McKay et al., 2022; Lenton et al., 2023; Rahmstorf, 2024; Wunderling et al., 2024), where runaway changes in the system can continue long after forcing stops once a threshold is crossed, often irreversibly; a softer example is risks with a long response time (like sea-level rises), where the system takes a long time to equilibrate to a given level of anthropogenic forcing, even in the absence of self-reinforcing tipping dynamics (Ditlevsen & Ditlevsen, 2023). In the final response ('saturate'), risk increases as a function of emissions but, after a certain point, ceases to increase, or may even decrease, as the risk may saturate beyond a certain point. An example of this would be risk from flooding in a coastal location resulting from sea level rise. Initially the risk of intermittent flooding may increase in line with sea level rise. However, if the sea level has risen past a certain point, the settlement may become permanently flooded, essentially saturating the risk.

3.3.3. Baseline

The risk *reductions* resulting from the various interventions must be measured against a baseline, so each climate risk management scenario requires the selection of a baseline trajectory**.** This is depicted in the figure as the top line, i.e., the envelope, of the risk profiles. In the absence of mitigation or adaptation, the baseline shape corresponds to the risk of loss and damage and is therefore dependent on a range of speculative factors, including, for example, time lags between cause and effect, the physical climate response, population changes, and technological shifts over time, amongst others (Newman & Noy, 2023). An example of a baseline would be a business-as-usual scenario.

There are four qualitatively distinct baselines for CO_2-forcing-equivalent emissions (Jenkins et al., 2018; Wigley, 1998) included in **Table 1**: (i) accelerating; (ii) linearly increasing; (iii) decelerating; and (iv) overshoot. The shape of the risk envelope for a scenario is obtained by first choosing a specific baseline and then applying the risk function discussed in the section above. As discussed above, the risk function is used to convert the quantity of a high-level factor (which might be expressed in, e.g. carbon dioxide emissions) into a comparable risk quantity.

3.3.4. Mitigation interventions

Decreasing climate risk relative to the baseline trajectory can be achieved through the deployment of the 'intervention' high-level factors: that is, mitigation, solar radiation modification and adaptation. The mitigation high-level factor represents the pathway that anthropogenic forcing takes over time. Mitigation wedges in the risk figure therefore correspond to interventions consisting of both emissions reductions and greenhouse gas removals, separated with a line through the wedge (reductions above and removals below). Greenhouse gas removal is chosen as the more general term over carbon dioxide removal to incorporate all contributors to global warming beyond just carbon dioxide. The top line of the mitigation wedges relates to a baseline risk trajectory, and the bottom line (in the absence of solar radiation modification deployment) corresponds to physical climate risk.

Table 1 includes three archetypal emission mitigation trajectories. In the 'risk increasing indefinitely' instance, mitigation interventions fall short of halting emissions, which results in cumulative global emissions following an indefinite upwards trajectory. This indefinitely increases risk. In 'net zero', net global emissions decrease to, and then remain at, net zero. 'Overshoot (net negative)' sees global emissions decrease to net zero and continue into net negative territory.

It is important to note that the diagrams are expressed in terms of *risk*, not *temperature* or *quantity* of emissions reductions and removals over time. Therefore, the wedge width for mitigation is proportional to the risk reductions at a given time resulting from historical activities relative to the baseline, which is a function of the emissions difference between the baseline emissions trajectory and the absolute level of emissions after mitigation in a given scenario. The specific wedge shape for mitigation interventions in a scenario is obtained using the risk function on the difference between the mitigation trajectory and baseline. For example, given a baseline of constantly increasing emissions, a mitigation state of net zero would represent a constantly increasing emissions difference. This would lead to an increasing temperature difference, and therefore an increasing risk difference. In particular, if the 'incremental' instance is chosen for the coupling between climate risk and anthropogenic forcing, a state of net zero would have a constant (horizontal) base of the mitigation wedge, and an ever-increasing top line for the baseline.

The shape of this emission trajectory wedge aligns with the research of all three working groups of the IPCC, with WGIII providing the assessment of mitigation levels in terms of emissions, and WGI and WGII calculating the temperatures from emission levels and indicating risk outcomes from temperature for the particular risk being considered (IPCC, 2022a).

3.3.5. Potential solar radiation modification deployment

Like the napkin diagram, the typology in **Table 1** incorporates potential risk reduction from solar radiation modification interventions. This high-level factor captures the temporal length of the solar radiation modification deployment, of which there are three instances: (i) 'none'; (ii) 'temporary'; and (iii) 'sustained'. In the temporary instance, a short-term period of solar radiation modification deployment is used to temporarily reduce temperatures in a given climate scenario. For example, it could be used to limit peak risk in an overshoot scenario, and then once warming-induced risk decreases to the desired level, solar radiation modification deployment would be ceased. In the sustained instance, continued deployment is used to keep temperatures, and therefore temperature-related risks, below certain thresholds on an ongoing basis. Where there is deployment, the level of physical climate risk lies at the bottom of the solar radiation modification wedge, as solar radiation modification activities affect the physical state of the climate.

Since the climate risk management scenarios are cast in terms of risk and not temperature, the shape of the solar radiation modification wedge in the case of a long-term deployment is more complex than the case of long-term sustained carbon dioxide removals. Whereas the risk reductions of one “unit” of carbon reductions and removals are ideally long lasting and therefore cumulative, the risk reductions of one “unit” of solar radiation modification are short-term and therefore dependent on the current and recent levels of solar radiation modification deployment. Therefore, termination of solar radiation modification deployment would cause the solar radiation modification wedge to quickly diminish, leading to the physical climate risk level rapidly rising to the base of the mitigation wedge, potentially driving responses more severe than if the physical climate risk level had more gradually risen to that level – this sudden change in the physical climate system resulting from termination of solar radiation modification is sometimes called “termination shock”. Therefore, while the risk reductions from solar radiation modification’s physical effects may keep increasing as the annual amount of solar radiation modification increases, the overall risk reductions may be attenuated owing the non-zero risk of termination shock; even without the probability of termination changing over time (which depends on governance), the magnitude of the impacts from termination would increase, and so the overall risk from termination shock could correspondingly increase. This tension means that the overall shape of the solar radiation modification risk reduction is nontrivial, contains much uncertainty, and should be considered with great care. Indeed, the requirement to consider the full gamut of risks associated with a particular course of action, rather than just an estimate of the likely temperature response, is a core benefit of using a risk-framed typology such as this. As with the baseline activities and mitigation interventions, obtaining the wedge shape of risk reduction for potential deployment of solar radiation modification must also incorporate use of the risk function to couple the deployment of solar radiation modification to the specific risk under consideration.

It should also be noted that solar radiation management may increase risk; for example, through disruptions to regional precipitation patterns, impacts on agricultural productivity, or the potential for geopolitical conflict (Bronsther & Xu, 2025; Sovacool et al., 2023; Wiener et al., 2025). Using it as a climate risk intervention thus requires serious consideration.

3.3.6. Adaptation interventions

Regardless of the speed and extent of mitigation interventions, the current level of carbon dioxide in the atmosphere means that some amount of warming is already occurring. Some form of adaptation interventions will thus be required. The risk reductions from adaptation interventions incorporate changes to learning over time as a result of improvements in both skill and capacity. The top line of the adaptation wedge corresponds to the level of physical climate risk, while the bottom line refers to the societal climate risk, or loss and damage. The size of the wedge refers to the actual risk reductions that occur as a result of the activities in question; it does not refer to the *amount* of adaptation required.

In **Table 1**, we present three stylistic instances of adaptation as a function of physical climate risk over time: (i) increasing, (ii) increasing and then decreasing, and (iii) constant. In the first instance ('increasing'), the community in question learns over time and is able to increase its adaptation levels, even in a world where physical climate risk remains constant or increases. In the second instance ('increasing then decreasing'), the community is initially able to adapt to low levels of climate risk, but over time is able to adapt less and less well, perhaps because exposure increases (e.g., due to a growing population), or owing to the existence of limits to adaptation as physical climate risk increases (IPCC, 2022b). In the final instance ('constant'), these factors balance out, and the risk reductions from adaptation as a function of physical climate risk remain constant over time; note this constancy need not correspond to a constant amount of adaptation activities.

3.4. Six Example Portfolios of Climate Interventions

To illustrate the value of the typology in exploring climate risk management scenarios, we present in **Figure 5** a case study of six scenarios that narratively highlight the potential utility of various interventions. These are presented at a global scale but can be customized to other spatial levels as necessary. The temporal scale here is considered to be over centuries, recognising that risks in the climate system have long timelines associated with them (IPCC, 2021).

All six of the example climate risk management scenarios here share the same risk type (here generalised climate risk), and risk thresholds (generalised thresholds: 'incremental', 'systemic', 'existential')**,** and the same risk function that couples a given intervention to its resulting risk reductions. The horizontal risk levels on the figures incorporate the concern that risk thresholds exist that once crossed may have qualitatively different impacts; since it may be of strategic interest to limit the physical climate risk below such thresholds, these levels are stylistically used as guiderails for the intended goal for different interventions. In addition, all six example scenarios here share the same baseline ('linearly increasing'), in order to focus discussions on different portfolios of interventions instead of speculative differences in baseline scenarios (Rogelj et al., 2023). Differences in these example scenarios therefore lie in the profiles of mitigation, solar radiation modification and adaptation interventions chosen, and the resulting levels of physical climate risk (after mitigation and solar radiation modification), and loss and damage risk (after adaptation).

Depending on the question one is interested in exploring, one could equally well choose different high-level factors to keep the same and/or different between each scenario. For example, one could construct scenarios that keep the mitigation and adaptation instances of the same for all scenarios (following Plan A below, for example), while changing the risk function that couples the intervention to risk type; this might be useful to explore how robust the outcomes of a particular set of climate interventions is to the range of possible climate responses.

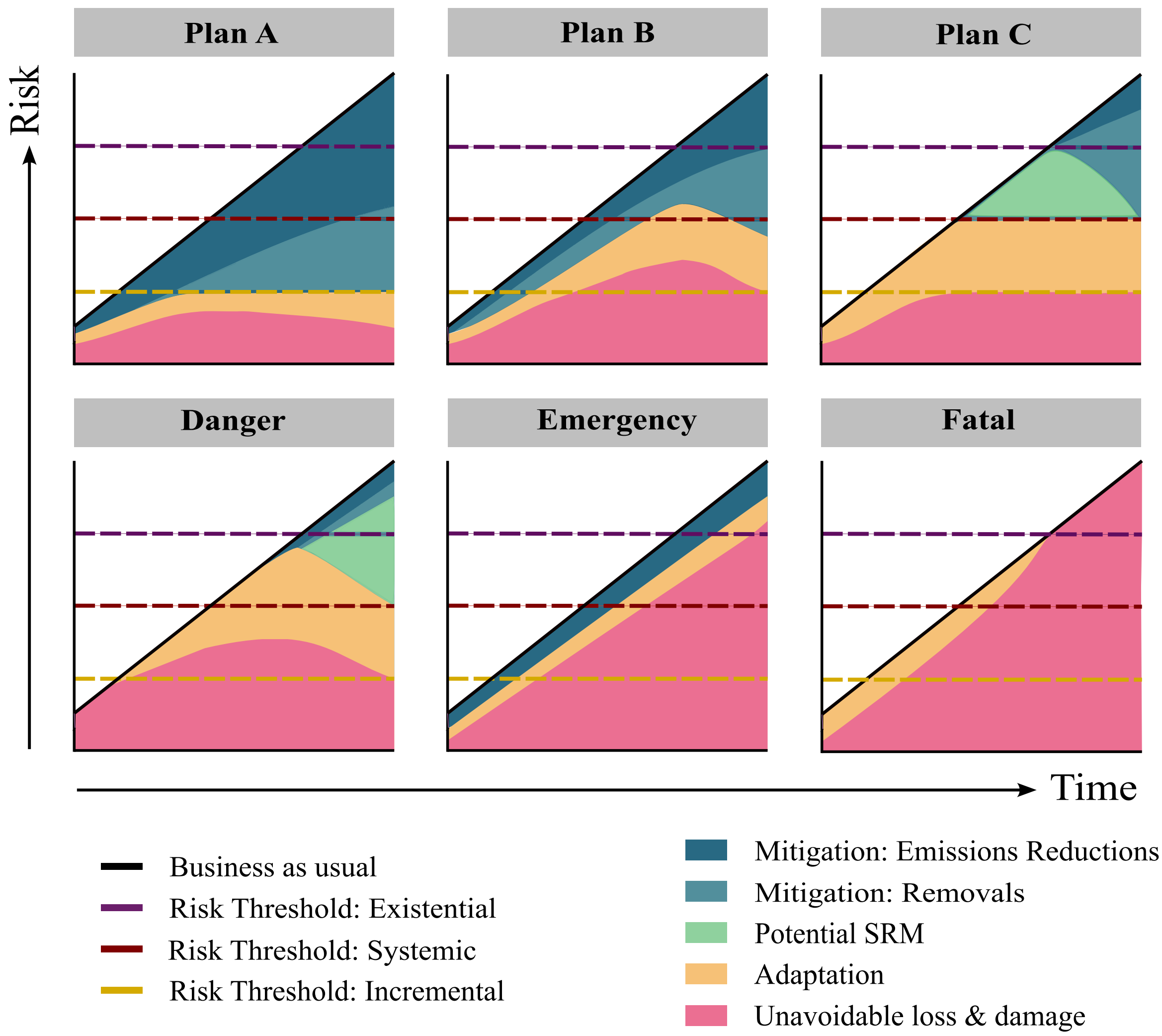


Figure 5: An illustrative set of six climate risk management scenarios. *To construct a scenario, a specific risk is chosen for the vertical axis, and any useful thresholds for this risk can be depicted with horizontal lines. The black line refers to the choice of baseline trajectory, and the wedges depict the risk reductions relative to this baseline resulting from various interventions, including mitigation (blue wedges), potential solar radiation modification (green wedge), adaptation (yellow wedge), and the remaining unavoidable loss and damage (pink wedge). Arbitrary sets of scenarios can be constructed using the typology introduced in this paper, depending on the specific risk being considered, and the sorts of integrated risk management scenarios that are being explored. Note that the variable on the vertical axis is risk, and therefore risk reduction, resulting from interventions is visualised – not the amount of an intervention used, or the temperature impact resulting from these interventions. The set of six scenarios presented here provide an example use of the typology, depicting the roles different interventions may play; as such the risk type, risk function, and thresholds are kept the same across each depicted scenario.*

Plan A is a stylised manifestation of a 'no overshoot' scenario. It incorporates early and rapid emissions reductions with levels of greenhouse gas removal limited to the necessary quantity to reach and sustain a stabilisation of risk over time. The simplest way that this might be achieved is if net zero emissions are reached and sustained, without net negative emissions after the point of net zero, and in an incremental physical climate response where risk increases with temperature and halts when it stabilises, and temperature is linearly proportional to cumulative emissions. In this stylised case, physical climate risk is stabilised at an 'incremental' level. The requirements of adaptation in this portfolio are therefore *relatively* low in comparison to other scenarios (but given the currently committed levels of warming and consequent climate impacts, any stabilisation at or beyond this state would crucially still entail high costs, whether monetary or otherwise). Thus, we chose the 'increasing' instance of adaptation, which results in a decrease over time in the risk of loss and damage.

In **Plan B,** slower initial ambition leads to temporarily overshooting the 'incremental' physical climate risk level, but a greater reliance on greenhouse gas removal enables net negative emissions and a consequent decrease in physical climate risk over time. In this example, the amount of risk reduction from adaptation activities is chosen from the typology to remain constant after the initial rise of physical climate risk (i.e. the wedge thickness is constant over time after the initial ramp up). This might be due to the more complex and less well-understood changes in the physical climate emerging from the both the unprecedented decreasing atmospheric carbon (from net negative emissions) and the fact that the physical climate risk state is constantly changing throughout the scenario. These factors might prevent learning that could increase the benefits from adaptation over time as the environment keeps changing. Despite the increase in the probability of crossing risk thresholds, loss and damage remains comparatively low due to the peak and decline in physical climate risk resulting from the mitigation efforts. Nevertheless, the risk of significant loss and damage still peaks above the incremental level, and in absolute terms would remain substantial.

In **Plan C**, mitigation goes far enough to both stabilise physical climate risk, and to begin decreasing it in an overshoot scenario; however, mitigation is not fast enough to prevent the peak physical climate risk level from first reaching the systemic risk level. Therefore, solar radiation modification is temporarily deployed to limit peak physical risk to a level that is more likely to be addressable through adaptation. Despite this, and unlike with the stabilisation portfolio in Plan A, we choose the 'constant' instance for adaptation. While the physical climate risk might have been stabilised due to the global temperatures being stabilised by solar radiation modification, the climate system will not be in a static equilibrium state as the varying levels of solar radiation modification changes (potentially unpredictably) the atmospheric composition and behaviour throughout deployment, and these changes may prevent adaptative capabilities from increasing over time. Further factors in the large-scale deployment of solar radiation modification could lead to further unintended side effects such as trans-border conflict over solar radiation modification deployment, which may also non-trivially affect the risk of loss and damage (including increasing it).

In **Plan D**, the "**Danger**" scenario, the trade-offs between solar radiation modification and mitigation-based on emissions reductions and removals are illustrated. Here, the risk reduction from mitigation relative to the baseline initially increases but after a certain period the reductions slow down, with emissions never reaching net zero and therefore never stabilising physical climate risk. Therefore, global-scale long-term solar radiation modification deployment must be constant and increasing to constrain physical climate risk and prevent it from indefinitely increasing.

In **Plan E**, the "**Emergency"** scenario, while mitigation efforts are pursued, the overall portfolio falls short of the actions required to constrain warming and therefore physical climate risk, leading to significant and permanent exceedance of risk boundaries. Adaptation efforts are also pursued but similarly fall short of the scale and effectiveness to constrain loss and damage, which therefore increases indefinitely.

In **Plan F,** the "**Fatal"** scenario, the limits to adaptation are highlighted (IPCC, 2022b). No additional mitigation efforts are pursued beyond what might have occurred in the absence of climate action (e.g. changing global emissions due to disruptions from escalating impacts of climate change). While adaptation is attempted beyond what might have occurred passively, we choose the 'increasing then decreasing' adaptation instance as the ever-increasing physical climate risk eventually surpasses humanity's ability to respond.

3.5. Customisation and further use of the typology

The typology presented in this article can be customized to represent other risks at varying spatial and temporal scales. An example is a coastal city government's consideration of the risks associated with sea-level rise. The city might use the typology to consider the issue at a high-level: business-as-usual emissions (the selected baseline) will result in sea-level rise, which might have impacts on infrastructure and livelihoods. Global and local emission reductions, as well as local adaptation measures, can decrease the risks that sea-level rise poses to the city. By visualising the risk reduction interventions together, the city would be able to identify whether local mitigation and adaptation interventions are working in tandem to decrease risk or are inadvertently in tension. The planned relocation of infrastructure as part of adaptation efforts may, for example, have an inadvertent impact of increasing travel distance to the new site, increasing local emissions.

The exercise might go a step further by including quantitative elements, like using emission projections to estimate the level of sea-level rise and thus to calculate the cost of adaptation measures. These estimations could be combined to arrive at an estimate of the cost of the unavoidable loss and damage associated with sea-level rise in the city. From this estimation, the city could, for example, better understand how the costs might be distributed.

The typology could also be used in conjunction with other risk tools and frameworks. For instance, as mentioned above, the burning embers tool (used by, e.g., the IPCC) can be used as the chosen risk. The tool could also, for example, be used in exploratory approaches commonly used by the decision making under deep uncertainty community (Workman et al., 2024), or other exploratory approaches like the adaptation pathways approaches (Werners et al., 2021), by providing structure around which to generate a variety of scenarios.

3.6. Conclusion

The current state of climate action indicates that overshooting the 1.5ºC temperature target is becoming increasingly likely. As such, a wider variety of interventions may need to be deployed, likely at increasing scale and under intensifying time pressure. Careful risk management will thus be required to avoid the inadvertent creation of additional risk from unintended side effects, yet risks accruing from the interactions between interventions can be difficult to gauge since approaches are often siloed.

In response, the climate risk management typology and example portfolios presented here provide a tool to holistically integrate climate interventions via their impacts on risk. Using this typology to create the portfolios brings together a variety of disciplines, and all IPCC working groups, in one figure; accordingly, the typology can contribute towards a common framework that supports better communication between practitioners and researchers. In particular, the six portfolios can act as a shorthand for the role of various climate interventions and the need to consider the overall risk reductions, and potential amplifications, resulting from each intervention. Further, the typology allows for arbitrary scenarios to be created for management of a variety of risks, across a range of timescales, localities, and sets of values. Importantly, it underlines the multifaceted approach to decreasing risk, which requires an integrative approach from decision-makers. This typology creates the structure for decision makers to do so.

It is important to emphasise that the interventions included in the typology and portfolios are selected in order to prioritise the understanding of potential linkages between them, rather than as an equally weighted endorsement of each, recognising the contested nature of some of these interventions in a variety of contexts. Indeed, one goal of integrating all options is to help in recognising trade-offs, more effectively use constrained resources, maximise synergies, and consider how the full gamut of implications and uncertainties relating to each intervention ultimately affects risk.

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Author contributions

CH contributed to conceptualisation, writing – draft, writing – review & editing. TW contributed to conceptualisation, writing – draft, writing – review & editing. EP contributed to conceptualisation, writing – review & editing. MO contributed to conceptualisation, writing – review & editing.

Competing interests

MO is one of three science advisors to the Climate Overshoot Commission.

4. Widespread misplacement of forestation projects in non-forest ecosystems

And the dead tree gives no shelter – T.S. Eliot, *The Waste Land*

The second research paper in the thesis presents the first of the two empirical studies. Forestation is a popular mitigation tool since aboveground carbon is comparatively easily measured. However, this may miss the broader ecological context, leading to forestation that is misaligned with the underlying ecosystem. This study is at a global level and quantifies the extent to which reforestation and afforestation projects are located in ecosystems where forests are ecologically inappropriate.

This chapter has not yet been submitted to an academic journal, although the intention is to do so shortly after PhD submission.

Widespread misplacement of forestation projects in non-forest ecosystems

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The proliferation of net zero targets has heightened interest in afforestation or reforestation projects to enhance carbon sequestration. However, concerns have been raised about the siting of these projects in non-forest ecosystems. Here we provide the first spatially explicit global assessment of the extent to which forestation sites overlap with natural areas of high tree cover or natural forest, triangulating the results using three independent frameworks. We overlay a published database of half a million forestation planting sites spanning 33 years with three independent maps of ecologically appropriate forest distribution (which move from broad ecosystem identity, through expected vegetation structure, to the environmental limits of closed forest). We found that more than half of global forestation area was located in areas consistently classified as non-forest under all three frameworks, potentially raising risks to biodiversity, the provision of ecosystem services, and long-term carbon storage. It is critical that greater scrutiny is applied to the siting of climate mitigation projects.

Key words: afforestation; forest potential; ecosystem suitability; carbon sequestration; climate mitigation

4.1. Introduction

Afforestation and reforestation (‘forestation’) are often used in pursuit of governmental and corporate climate targets to offset emissions (Holl & Brancalion, 2020; S. Smith et al., 2024). Reaching net zero will likely require large-scale carbon dioxide removal (IPCC, 2023b), of which forestation projects presently constitute the dominant form (S. Smith et al., 2024). As such, current pathways for achieving the Paris Agreement’s temperature goals will depend in part on the integrity of forestation projects. This reliance means that it is imperative that carbon is durably stored, consistent with the assumptions embedded in many integrated assessment models (Brunner et al., 2024). However, the risks of reversal for carbon sequestered during forestation projects – whether from land-use change or natural disturbances – are increasingly highlighted (Anderegg et al., 2020; Fuss et al., 2018; Streck et al., 2026). A key reversal risk is the extent to which forestation occurs in non-forest ecosystems, given that natural disturbances and constraints may inhibit the durability of the forestation project. A growing body of work is raising concerns about the siting of forestation projects in ecosystems that may not support forests (Bond et al., 2019; Enríquez-de-Salamanca, 2024; Luedeling et al., 2019; Stevens & Bond, 2024). However, the extent of the overlap of forestation efforts with non-forest ecosystems has not yet been quantified.

Tree planting projects can occur in areas that previously supported tree-dominated ecosystems (reforestation) or that have long been non-tree-dominated (afforestation) (Andres et al., 2023). In both cases, trees are being introduced into treeless systems, which can be so due to at least three reasons. First, some systems were historically forested but have been deforested due to anthropogenic or natural drivers (Balboni et al., 2023; Haddad et al., 2024). Second, in some locations, trees may not be able to establish or persist as a result of environmental constraints, such as flooding, nutrient limitation, or shallow soils (Bond, 2008; Kozlowski, 1997). Third, in some areas, the climate may be suitable to support a forest, but the ecosystem remains open due to a co-evolution with disturbance regimes, particularly fire and herbivory (Bond, 2019; Staver et al., 2011). As such, in these systems, disturbance regimes have influenced – and continue to influence – biodiversity, structure, and function, thus maintaining an open, lower-biomass state over long timescales rather than allowing a transition to forest (Pausas & Bond, 2020).

We follow the convention of Pausas & Bond (2020) in referring to these systems as ‘open’ ecosystems, incorporating typical examples like savannas, shrublands or open woodlands. Open ecosystems are not degraded forests but rather ancient, biodiverse systems with distinct ecological identities (Pausas & Bond, 2019; Veldman, Buisson, et al., 2015). This classification system is driven by the need to capture inappropriate forestation in African open ecosystems such as grasslands and savannas, where "woodland" typically refers to scattered trees or open woodland ecosystems that support a light-loving understory. Different classification systems are in use in other regions; e.g. in Europe and North America, "woodland" can equate to native closed canopy forest. These differences will be addressed in future work.

Misidentifying the type of treeless system has implications for not only the durability of carbon storage in practice, but a range of ecosystem services supplied by open, non-forest ecosystems. In these ecosystems, disturbance regimes and environmental constraints may limit growth or lead to mortality of planted trees, thereby compromising a central tenet of voluntary carbon markets: permanence of carbon storage. In semi-arid and grassy systems, for example, afforestation has been linked to declines in water availability and alterations to fire regimes, both of which may further reduce the durability of carbon stocks (Cao et al., 2011; Honda & Durigan, 2016; Mátyás et al., 2013; Nosetto et al., 2012; Stevens & Bond, 2024). Afforestation may also impact non-forest ecosystems more broadly, with consequences for biodiversity and ecosystem functioning (Prangel et al., 2023). It may also reduce the provision of key ecosystem services, including grazing livestock production, grassland-dependent pollination services to pollinated crops, as well as cultural services such as the availability of medicinal plants (Briske et al., 2024; Luvuno et al., 2022; Skhosana et al., 2025; Ullah et al., 2025; Vetter, 2020). These potential impacts highlight the importance of identifying ecologically appropriate areas for forestation rather than those that may be appropriate only on some metrics (e.g., climate) but not all.

However, distinguishing between the different types of treeless systems is not trivial to do at a global scale. For example, at broad spatial scales, the different systems can present a similar structure, such that grasslands, savannas, and degraded forests may appear indistinguishable using widely applied remote sensing methods (Ratnam et al., 2011; Veldman, Overbeck, et al., 2015b). This is compounded by limitations in what these approaches can detect. Thus, short or widely spaced trees, which are common in many savannas, may not be captured in remote sensing data (Brandt et al., 2020). Grasses and other low vegetation are also not consistently mapped, and satellites cannot easily detect vegetation beneath tree canopies (Gao et al., 2020). This has the same effect: that is, the different systems appear structurally similar at a global scale. At the same time, the environmental constraints that limit tree growth are often local in character, and are therefore not well represented in global datasets that operate at coarse resolutions (Markham et al., 2023; Zhang et al., 2023). Disturbance regimes, too, are not well captured at a global scale, particularly since fire regimes can be spatially and temporally variable (Andela et al., 2017; Archibald et al., 2013). Accordingly, global analyses may struggle to distinguish between different types of treeless systems.

Further, a key challenge in making this distinction is that there is no single ecological definition of forest. Instead, forests are defined using a range of criteria, including percentage tree cover, canopy height, and land use designation (Chazdon et al., 2016; Food and Agriculture Organization of the United Nations, 2018). Definitions vary across institutions and according to their intended use (Chazdon et al., 2016). Many widely used definitions that rely on structural thresholds, such as minimum canopy cover or tree height, do not account for the ecological processes, like those that maintain open ecosystems (Sexton et al., 2016; Veldman, Overbeck, et al., 2015a). In particular, approaches that are based primarily on environmental constraints may successfully estimate where trees could occur under suitable climate and soils, but can overestimate forest potential where disturbance processes such as fire and herbivory limit tree growth or persistence (Veldman, Overbeck, et al., 2015a). As a result, areas that are climatically suitable for forests, but are maintained as open systems by disturbance or environmental constraints, may still be classified as forest or as suitable for forestation (Pausas

& Bond, 2020). The lack of a common definition increases the risk of misclassification. Indeed, a number of large global forest potential maps – which are often used to guide the planning of the location of forestation sites – have been criticised for insufficiently incorporating ecological constraints like water availability and disturbance regimes (Krause et al., 2019; Parr et al., 2024; Veldman et al., 2019; Veldman, Overbeck, et al., 2015a). As a result, these approaches may overestimate the extent of land suitable for forestation, particularly in open and dryland systems.

To address this challenge, we explicitly integrate three complementary ecological frameworks that represent these distinct perspectives. First, we use classified ecoregions to characterise broad ecosystem identity, distinguishing forest, open, mosaic, and treeless systems based on vegetation structure and evolutionary history (Dinerstein et al., 2017). This provides a coarse, ecoregion-level context for where planting occurs, although it does not resolve fine-scale heterogeneity, as forested ecoregions may contain open landscape patches and open ecoregions may include patches of tree cover. Second, we use a natural land cover model from Bastin et al. (2025) to estimate expected tree, short vegetation, and bare ground cover under environmental filtering and disturbance regimes such as fire and herbivory. This provides a continuous representation of vegetation structure and helps to account for within-biome variability. Third, we use a forest potential map from Fesenmyer et al. (2025) as a more conservative estimate of where closed-canopy forest is environmentally feasible, accounting for key ecological constraints. This identifies areas where sustained forest cover is more likely to be achieved and thus represents more robust opportunities for reforestation.

These frameworks provide a hierarchical view of vegetation, moving from broad ecosystem identity, through expected vegetation structure, to the environmental limits of closed forest. We therefore use agreement across frameworks as an indicator of ecological appropriateness. Where all three agree on an area being forest-dominated, there is strong evidence that forestation is placed within an appropriate system. Where none of the frameworks support forest-dominated ecosystems, this indicates landscapes in which vegetation structure may be shaped by disturbance or other processes rather than climate alone, and where the appropriateness of forestation is less certain. In these cases, forestation may be misaligned with underlying ecosystem dynamics.

Using these frameworks, we provide a global, spatially explicit assessment of where forestation projects are located relative to forest-dominated systems. We ask three questions: (i) how much forestation is occurring in areas consistently identified as forest-dominated across frameworks, at both global and continental scales? (ii) To what extent do different frameworks agree or disagree on where forestation is ecologically appropriate? (iii) Are misplaced forestation projects disproportionately located in particular ecological contexts, especially in open and disturbance-maintained systems like savannas?

Based on these questions, we expect that a substantial proportion of forestation occurs outside areas consistently identified as forest-dominated across frameworks, given that previous work has highlighted that global forestation and restoration efforts may be directed towards non-forest ecosystems (Bond et al., 2019; Fleischman et al., 2020; Parr et al., 2024; Veldman et al., 2019). We also expect limited agreement between frameworks, reflecting the different processes they capture. In particular, approaches based primarily on biophysical constraints are likely to indicate broader areas of potential tree cover, while more conservative estimates that account for disturbance and other constraints will identify fewer areas as suitable for sustained forest (Fagan, 2020; Pausas & Bond, 2020). Finally, we expect that any misplacement will be concentrated in open and disturbance-maintained systems, especially savannas, where climates are suitable for tree growth, but vegetation structure is maintained by disturbance. These systems are frequently misclassified as degraded forests (Pausas & Bond, 2019, 2020), and are often more accessible and easier to plant than areas that would naturally be closed forests, increasing the likelihood that forestation is concentrated in these landscapes (Bond et al., 2019; Fagan, 2020; Veldman, Overbeck, et al., 2015b; Veldman et al., 2019).

While previous work has highlighted that forest potential maps may identify areas for forestation that are not ecologically appropriate, and that forestation commitments may exceed feasible forest area (Bond et al., 2019; Fleischman et al., 2020; Parr et al., 2024; Veldman et al., 2019), the extent to which these issues are already reflected in the spatial distribution of forestation projects remains unclear. By integrating multiple ecological frameworks, we offer a more nuanced evaluation of where tree planting is likely to be ecologically appropriate and where it may be misaligned with underlying ecosystem dynamics. Quantifying the extent of such misplacement is critical not only for understanding local ecological impacts, but also for assessing the durability and resilience of carbon storage within nature-based climate mitigation strategies.

4.2. Methods

We quantified the ecological context of global forestation by overlaying a spatial database of planting sites with three complementary ecological frameworks representing ecosystem identity, expected vegetation structure, and forest suitability. These three frameworks provide distinct but complementary perspectives on where forestation is suitable.

All spatial operations, analyses, and visualisations were performed in RStudio (R version 4.5.1; RStudio version 2025.09.0), using the *tidyverse, sf, rnaturalearth, arrow, terra, exactextractr*, *biscale* and *cowplot* packages (Baston, 2024; Hijmans et al., 2026; Massicotte & South, 2025; Pebesma, 2018; Posit team, 2025; Prener et al., 2025; Richardson et al., 2026; Wickham et al., 2019; Wilke, 2025). All spatial overlays and area calculations were performed in a global equal-area coordinate reference system (EPSG:6933).

4.2.1. Forestation

The forestation dataset is derived from John et al. (2025) (available at: https://doi.org/10.7910/DVN/ZJODGO). The original dataset contains 1,289,068 afforestation and reforestation planting sites from 45,628 projects over 33 years, which were retrieved from eighteen web databases.

To ensure that the dataset was limited to terrestrial forestation projects as well as projects intended to increase forest cover, we screened the provided project descriptions for two groups of keywords: (i) non-forest terrestrial vegetation ("savanna", "savannah", "grassland", "prairie", "steppe", "rangeland", "scrubland", "shrubland", "thicket", "woodland(s)"); and (ii) blue carbon or marine vegetation ("kelp", "seaweed", "seagrass", "saltmarsh" / "salt marsh", "blue carbon" (with or without a hyphen), "ocean restoration", "coral", "reef"). We used the *grepl* function to run a case-insensitive search to remove projects that had descriptions containing one of the relevant keywords. This step was necessary because some projects captured by this dataset, labelled as "reforestation" or "restoration" in open ecosystems, may instead aim to restore historically non-forested vegetation, which does not necessarily involve an increase in forest cover. Including such projects would risk conflating forestation with ecologically appropriate restoration of open ecosystems. As noted above, removing "woodland" is important in African ecosystems but may inadvertently remove areas suitable for reforestation in other regions where the term "woodland" can equate to closed canopy forest. This will be addressed in future work.

We applied a series of filtering steps to remove clearly erroneous non-terrestrial records. Sites located in Antarctica, or with undefined or zero area after reprojection were excluded. In addition, sites that were explicitly labelled in the original dataset as having boundaries that closely resembled administrative regions (e.g. countries or provinces) were excluded, as these were identified by the dataset authors as being either unrealistically large or not clearly enough delineated to just the project area. We also excluded extremely large polygons (> 50,000km^2) based on inspection of the largest 1% of remaining planting sites by area (~8970 sites). We first inspected all sites larger than 50,000km^2 (~25 sites), then examined an additional ~50 immediately below this threshold. We then randomly sampled the remaining projects until ~200 sites had been reviewed in total. Based on this assessment, 50,000km^2 was identified as a reasonable threshold for excluding anomalously large polygons. See **Table S 1** for a list of planting sites that were excluded after manual inspection.

To avoid double-counting land area, nested subset polygons were removed from area-based analyses (leaving 'unique footprint sites') while all records were retained for analyses based on project counts ('planting site records'). The original forestation dataset included a *nested_in* field identifying overlapping sites. Polygons with >95% overlap in the area of both sites were classified as duplicates by the original dataset authors, whereas polygons with >95% of their area contained within a larger polygon were classified as subsets. We retained the larger polygons since they were intended to represent the full project footprint, and excluded the nested subsets from area-based analyses. In the original dataset, subset polygons occurred predominantly within smaller project footprints (<500km^2), so we did not expect this choice to have a substantial effect on the spatial precision of our analyses.

We were left with a dataset of 886,014 planting site records, with 491,069 unique footprint sites. See **Table S 2** for details on how many planting sites were excluded by category.

4.2.2. Quantifying extent of placement in forest-dominated ecosystems

The cleaned forestation database was compared with three frameworks for delineating forest-dominated ecosystems (**Table 2**). For each planting site polygon, we calculated the proportion of area overlapping forest-dominated areas under each ecological framework. A planting site was classified as placed within a framework if ≥50% of its area overlapped the corresponding forest layer (viz. areas of biophysical forest potential, areas with predicted natural tree cover

≥60%, or ecoregions classified as forest). Sites with <50% overlap were classified as misplaced. The same classification rule was applied across all three frameworks. After classification, we summed the area of all correctly placed sites to calculate the proportion of total forestation area correctly situated within each framework. Because the natural vegetation model is coarse (~25 km), results should be interpreted as reflecting landscape-scale ecological context rather than local suitability.

We assessed sensitivity to the 50% threshold by repeating the analysis using alternative placement thresholds ranging from 0.1 to 0.9. Our results were highly insensitive to threshold choice, with each of the frameworks reporting changes of less than 1% across all thresholds (**Table S 3**).

Table 2: Ecological frameworks used. *Each of these frameworks is explained in further detail in the following subsections.*

Framework	Description	Source
Ecoregion-based forest classification	Ecoregions classified into forest, open, mosaic, and treeless systems to represent broad ecosystem identity based on vegetation structure and evolutionary history; provides coarse context for ecoregion-level patterns.	Manual classification of RESOLVE ecoregions
Predicted natural tree cover	Continuous model of expected tree, short vegetation, and bare ground cover under environmental filtering and disturbance (e.g. fire, herbivory), providing a representation of underlying vegetation structure; forest defined as ≥60% predicted tree cover.	Bastin et al. (2025)
Biophysical forest potential	Binary map of where closed-canopy forest is environmentally feasible under biophysical constraints, accounting for disturbance and other ecological limitations; represents conservative estimates of sustained forest cover.	Fesenmyer et al. (2025)

In addition to the overlap analysis, we extracted summary statistics describing the ecological context of forestation sites, including mean predicted natural vegetation cover and the distribution of forestation area across ecoregion categories.

We evaluated forestation sites against three complementary ecological frameworks capturing ecosystem identity, vegetation structure, and forest feasibility. The predicted tree cover and biophysical forest potential layers represent distinct perspectives on forest potential. The predicted tree cover model provides a continuous estimate of vegetation structure under environmental filtering and disturbance, while the biophysical forest potential map applies more conservative ecological and socio-economic constraints to identify areas where closed forest is feasible. Using both allows us to assess how conclusions depend on different assumptions about forest potential and to identify areas of agreement and disagreement between approaches.

4.2.3. Ecoregion-based forest classification

The ecoregion-based forest classification uses the RESOLVE Ecoregions 2017 database (https://ecoregions.world/), which contains 847 terrestrial ecoregions and is often used to represent ecosystem diversity in conservation and international agreements (Dinerstein et al., 2017, 2019; Olson et al., 2001; J. R. Smith et al., 2018, 2020).

We classified the ecoregions into forest and non-forest categories depending on canopy structure. Each of the 847 ecoregions was classified manually through review of descriptions of its vegetation structure and ecology. Forest ecoregions were defined as those where closed-canopy forest is extensive, characterised by tree-dominated systems with a closed (or nearly closed) canopy that shades out a persistent understory of grasses or shrubs (Bond, 2019; Bond & Parr, 2010). This definition includes both terrestrial and mangrove forests (Alongi, 2009). Non-forest ecoregions were further divided into three sub-categories: forest-non-forest mosaic, open, and treeless biomes. Mosaic ecoregions contain spatial mixtures of forest and non-forest ecosystems, with closed-canopy forest occurring in restricted patches within a broader landscape of other ecosystem types. An example of this is a forest-savanna mosaic (Charles-Dominique et al., 2018). Open ecoregions are tree-dominated systems that lack extensive closed canopy and typically support a persistent grassy understory, with vegetation structure often maintained by disturbance regimes such as fire or herbivory (e.g., savannas and woodlands) (Pausas & Bond, 2020). Treeless ecoregions are those where conditions (e.g.

climatic or edaphic) preclude persistent tree cover. Examples include deserts, wetlands, flooded grasslands, or tundra. The full classification of ecoregions is provided in **Table S 4**.

We note that our definitions rely on structural distinctions between closed-canopy forest and more open, tree-dominated systems. In particular, some ecosystems classified here as “open” (e.g. woodlands or woody savannas) may be considered “forests” under alternative definitions based on tree-cover thresholds. Our classification therefore reflects a more conservative definition of forest, emphasising closed-canopy systems.

4.2.4. Predicted natural tree cover

Predicted natural tree cover was derived from the natural vegetation model of Bastin et al. (2025), which is available at https://github.com/Nicolas-Latte/GANVC. In short, the model estimates fractional natural land cover (tree, short vegetation, and bare ground) at a spatial resolution of 0.25° (~25 km at the equator), using land cover data from protected areas, as well as environmental determinants.

The model, based on neural network approaches, was trained on photo-interpretations of 40,520 0.5-hectare plots from protected areas globally, so chosen as to simulate natural land cover with limited human interference. This training data was augmented by environmental determinants (climatic, edaphic, topographic, fire, and wildlife herbivory) to predict fractional land cover. These determinants contained the following data: the climate variable contained temperature and precipitation data; edaphic incorporated soil organic carbon stock and sand content (both from 0-to-15 cm), as well as depth to bedrock, elevation, and hillshade; fire frequency concerned burned-area and quality; while the herbivory variable comprised the global distribution of wild mammal herbivore biomass (Bastin et al., 2025).

We used a threshold of ≥60% predicted natural tree cover to delineate forest consistent with closed or near-closed canopy conditions. This threshold is also consistent with the threshold used by Fesenmyer *et al.* (2025) in constructing their forest potential map. We assessed sensitivity to this threshold by varying it from 10-90% in ten-percentage-point increments (***Table S 5Error! Reference source not found.***) and found that the percentage of projects that fell in areas identified as forest-dominated under the framework declined smoothly as the tree-cover threshold increased. The percentage of projects decreased from 72% at a 30% threshold to 13% at 80% tree cover, with no abrupt changes. Our chosen tree cover threshold (60%) had an intermediate value of 33%, suggesting that results are not driven by a specific cutoff.

4.2.5. Biophysical forest potential

The biophysical forest potential dataset is sourced from Fesenmyer et al. (2025) and is available at https://doi.org/10.6084/m9.figshare.27335799. This framework provides a conservative estimate of forest feasibility (as areas that could support 60% or more tree cover) rather than a definitive classification. The dataset predicts forest potential at a spatial resolution of 1km, represented as a binary forest potential surface with values of 0 (no potential) or 1 (potential). It was generated by finding the agreement between the forest potential maps of Bastin et al. (2019), filtered to 60% potential tree cover, and Walker et al. (2022), filtered to closed forest. The Bastin et al. (2019) forest potential map was created to estimate natural tree cover and used 78,774 photo-interpretations of protected areas to estimate tree cover using climate, edaphic, and topographic variables. The Walker et al. (2022) map identifies current and potential carbon storage in above-ground woody biomass, separating out closed forests, open forests, and nonwoody systems using a carbon threshold. After combing these two datasets to find areas of agreement, Fesenmyer et al. (2025) filtered the resulting dataset to exclude areas with two or more non-cropland fires between 2002 and 2022 that were not identified as being cropland or oil palm plantation in 2022. We used the base forest potential map rather than the scenario-specific layers (e.g. ‘high individual rights scenario’), which were predicated on a ‘maximum reforestation potential’ scenario that excludes all existing forest. Given that our forestation database spans 30 years, older projects may now be situated in areas classified as 'existing forest'. Using the maximum reforestation potential map (and the associated scenarios) risks misclassification of the planting sites as falling outside of forest potential areas.

4.3. Results

We assessed forestation placement in forest-dominated ecosystem, using three ecological frameworks that vary in conservativeness, from broad ecoregion classifications to more restrictive biophysical forest potential.

4.3.1. Frameworks agree on substantial misplacement of forestation projects

Globally, only 14.7% (~ 60.6 Mha) of forestation area was consistently classified as forest-dominated across all three frameworks (**Figure 6**). In contrast, a majority (53.8%, or ~ 221.7 Mha) is consistently located in ecosystems that are not classified as forest under any framework. The remaining 31.4% of forestation area (~ 129.5 Mha) occurred in regions where frameworks disagreed (i.e. classified as forest by one or two frameworks only).

Most forestation area with partial agreement occurred where the predicted natural tree cover and ecoregion frameworks both classified sites as forest (15.1%, or ~ 62.4 Mha). In contrast, combinations involving the biophysical forest potential framework were rare, accounting for only 0.48% (~ 2 Mha) with ecoregions and 0.03% (~ 0.14 Mha) with predicted natural tree cover.

Cases where only a single framework classified forestation as occurring in forest were also dominated by the broader classifications. Forestation classified as forest only under the ecoregion framework accounted for 10.8% (~ 44.7 Mha) of total area, while 4.9% (~ 20.3 Mha) was classified as forest only under predicted natural tree cover. In contrast, forestation classified as forest only under biophysical forest potential was negligible (0.001%, ~ 0.005 Mha).

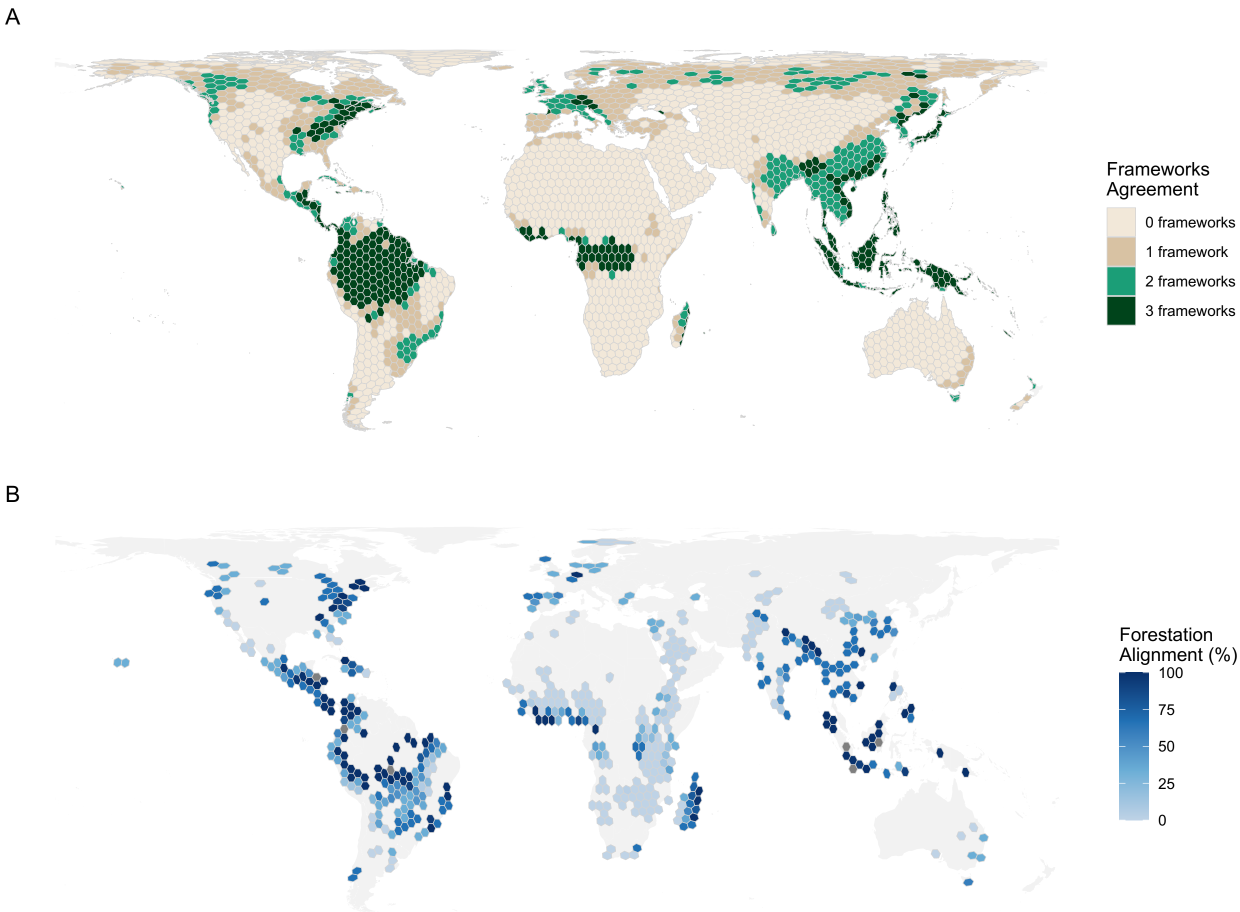

***Figure 6: Global patterns of agreement and placement of forestation projects relative to ecological frameworks**. Panels show (A) the number of ecological frameworks (forest ecoregions, predicted natural tree cover, and biophysical forest potential) that support forestation within each hexagonal grid cell (0–3). Areas shown in very light brown indicate locations where none of the frameworks support forest-dominated ecosystems, while dark green indicates agreement across all three frameworks that the relevant forestation area is located in a forest-dominated ecosystem. (B) Percentage of forestation area within each hexagonal grid cell that is placed within forest-dominated systems across all frameworks. Higher values (dark blue) indicate greater placement within forest-dominated systems, while lower values (light blue) indicate placement outside these systems. The grey indicates areas where there are no forestation projects. Maps depicting each individual framework's classification of forestation as being in forest or non-forest-dominated ecosystems are in **Figure S 1**.*

When considered independently, the three frameworks differ in the extent to which forestation is classified as occurring in forest-dominated systems globally, with 17.3% under biophysical forest potential, 33.3% under ecoregions, and 35.5% under predicted natural tree cover (**Figure S 2**).

4.3.2. Continental heterogeneity

The extent of agreement differed noticeably across continents (**Figure 6**), with clear differences in both the extent and make-up of agreement (**Figure 7**). This heterogeneity was also evident at the individual framework level (**Figure S 3**).

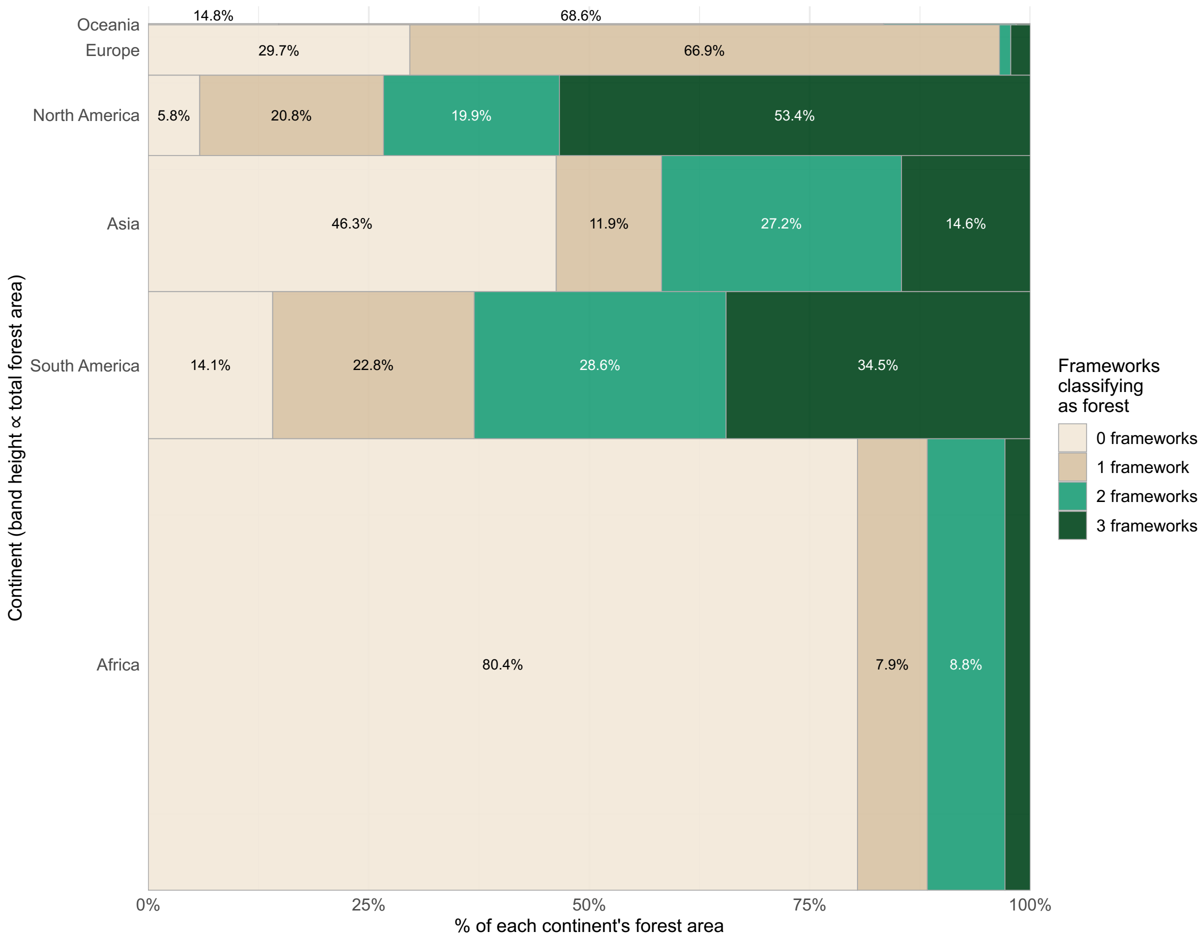

Figure 7: Continental variation in the number of ecological frameworks classifying forestation as occurring in forest-dominated systems. *Bars show the percentage of total forestation area within each continent classified as forest by zero to three frameworks (ecoregions, predicted natural tree cover, and biophysical forest potential). The vertical thickness of each bar is proportional to the total forestation area of that continent. Oceania makes up a very small share of total forestation area and is therefore only minimally visible at this scale. Values of 0 indicate that none of the frameworks classify forestation as occurring in forest, while values of 3 indicate agreement across all frameworks. Values less than 5% are not labelled for clarity.*

Africa represents the most extreme case of ecological misalignment globally, with over 80.4% (~ 172.6 Mha) forestation occurring in systems that are consistently identified as non-forest across all frameworks. Only 2.9% (~ 6.1 Mha) of forestation area was consistently classified as forest across all frameworks. Partial agreement was limited and occurred primarily between the predicted natural tree cover and ecoregion frameworks (8.6%).

In contrast, more than half of North American forestation area (53.4%; ~ 20.4 Mha) was classified as forest across all three frameworks, and only 5.8% similarly classified as non-forest under all frameworks. Partial agreement occurred across both single- and two-framework classifications, again dominated by combinations of predicted natural tree cover and ecoregions.

Just over a third (34.5%; ~ 24.1 Mha) of South American forestation area was classified as forest across all frameworks, while 14.1% (~ 9.9 Mha) was classified as non-forest. A sizeable proportion of forestation area (28.6%) was classified as forest by two frameworks, primarily the predicted natural tree cover and ecoregion frameworks.

Asia also showed substantial areas consistently classified as non-forest, with 46.3% (~ 29.9 Mha) of forestation area classified as non-forest under all frameworks. However, a larger share of forestation area than in Africa was classified as forest by two frameworks (27.2%) or all three frameworks (14.6%), again dominated by agreement between predicted natural tree cover and ecoregions.

Europe and Oceania were both characterised by dominance of single-framework classifications. In Europe, 66.9% (~ 16.0 Mha) of forestation area was classified as forest under only one framework, almost entirely driven by the ecoregion framework, while a further 29.7% was classified as non-forest under all frameworks. Similarly, in Oceania, 68.6% (~ 0.37 Mha) of forestation area was classified as forest under only one framework, again primarily under ecoregions, while only 1.6% was classified as forest across all three frameworks.

Across all continents, agreement was primarily driven by the predicted natural tree cover and ecoregion frameworks, with combinations involving biophysical forest potential remaining rare.

4.3.3. Ecological context of forestation

Looking more closely at the ecological contexts of the forestation sites, the predicted natural vegetation model shows that the median predicted tree cover across all sites globally was 54% (IQR 33-74) (**Table S 7**), which falls short of the 60% tree-cover threshold that we used to define forest-dominated ecosystems. The median for short vegetation (grasses and shrubs) was 36% (IQR 25-40). The median predicted tree cover across all African forestation sites was lower than our threshold of 0.6 (at 0.53) with a relatively high median predicted short vegetation (0.45) (**Table S 7**).

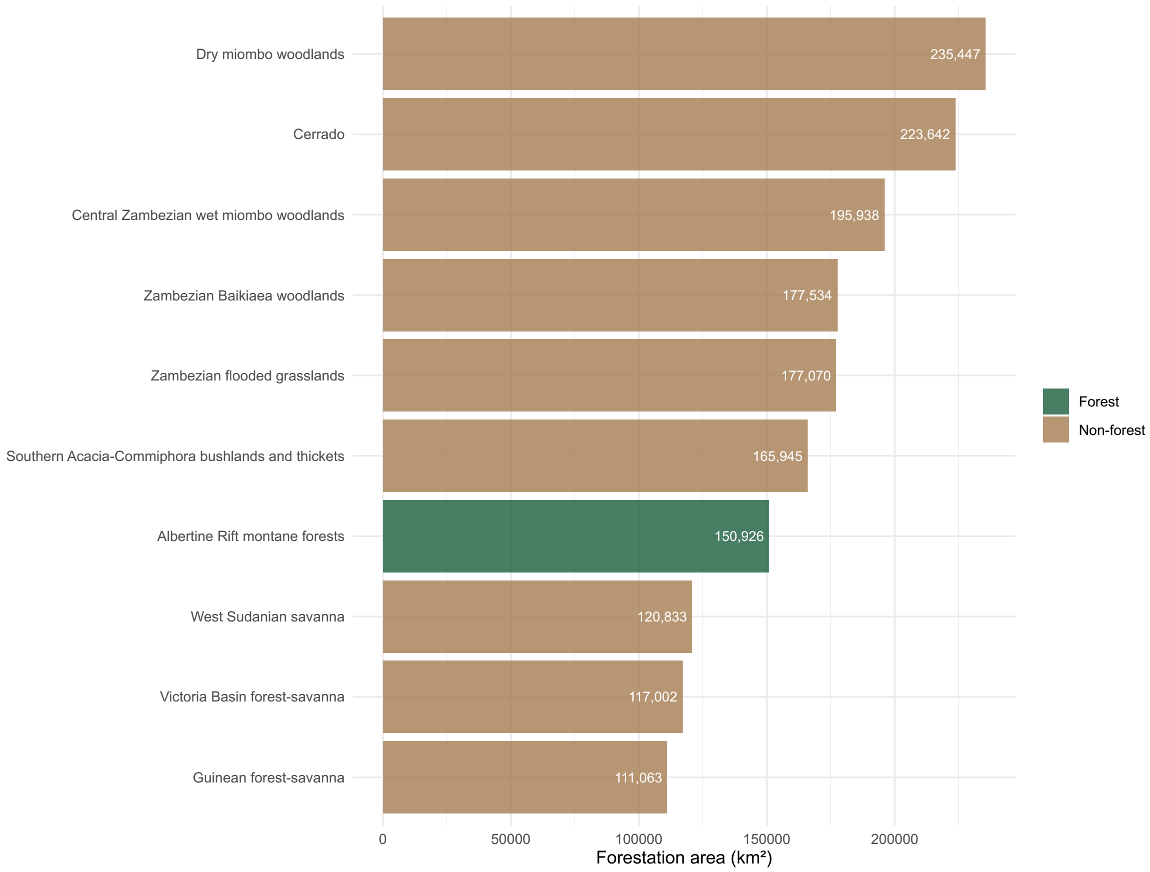

Figure 8: Top 10 ecoregions by total forestation site area. *Bars and values show the total area (km^2) of forestation sites located within ecoregions not classified as forest. Ecoregions are classified as forest or non-forest, with colours indicating within which category the ecoregion falls (gold for non-forest and green for forest).*

Under the ecoregion-based classification, nearly half (45.4%) of forestation area is located in non-forest ecoregions, with 30.7% in open, 8.3% in treeless, and 6.5% in mosaic ecoregions. Eight of the ten ecoregions containing the greatest forestation area are classified as non-forest ecosystems (**Figure 8**).

Only 14.3% of African forestation area occurs within forest ecoregions, while 39.9% occurs in open, 11.5% in mosaic, and 8.3% in treeless ecosystems. Of the 10 ecoregions that had the largest area of forestation, nine were in Africa, with five of that number in open ecosystems

(**Figure 8**; full list in **Table S 8**). Under the ecosystem-based classification, 33% of the forestation in South America fell in open ecosystems, with the Cerrado ecoregion having the second largest amount of forestation (~22 Mha) located within a single ecoregion globally (**Figure 8**). Three Asian ecoregions also contain the highest numbers of individual project sites, including the treeless ecoregion Southeast Tibet shrublands and meadows (105,040 sites), as well as two forest ecoregions: Changjiang Plain evergreen forests (58,414 sites), and Central China Loess Plateau mixed forests (50,626 sites) (**Table S 8**). The majority of Oceanian forestation area fell within two forest ecoregions: Eastern Australian temperate forests (0.34 Mha) and Tasmanian temperate rain forests (0.10 Mha) (**Table S 8**).

4.4. Discussion

Our analysis reveals a concerning mismatch between global forestation efforts and ecological context. Across three independent frameworks, the majority (53.8%, ~ 221.7 Mha) of forestation occurs outside forest-dominated systems, with over half of all mapped projects consistently located in ecosystems where forest is not the dominant or stable vegetation form. This pattern reflects strong cross-framework agreement that forestation is systematically occurring in non-forest environments

Results differed considerably across continents, with particularly high levels of misplacement in Africa (80.4%) while North America had over half of its forestation area located in ecologically appropriate forest-dominated areas. Open ecosystems, like savannas and open woodlands, received the highest share of forestation in non-forest ecosystems, with almost a third (30.7%) of global forestation projects and nearly 40% of African forestation area. These findings indicate that forestation frequently occurs outside ecosystems naturally dominated by closed-canopy forest, although the ecological appropriateness of individual projects depends on local context.

4.4.1. Widespread ecological misplacement

Across all three frameworks, we find that forestation is predominantly located outside forest-dominated ecosystems. Only a small proportion of forestation area is consistently classified as forest across all frameworks, while a majority falls outside forest ecosystems under all three, indicating widespread ecological misplacement. A growing body of literature raises concerns that forestation may be occurring in non-forest ecosystems (Bonanomi et al., 2019; Bond et al., 2019; Cao et al., 2011; Fernandes et al., 2016; Luedeling et al., 2019; Scogings, 2023; Veldman, Overbeck, et al., 2015b, 2015a; Vetter, 2020). Our results provide global spatially explicit evidence that this is the case.

This may be driven by methodological choices, which may have excluded some ecologically suitable areas; for example, areas in the UK where woodland refers to native forest and tree planting is suitable. Although the frameworks differ in their classification of approximately one-third of global forestation area, they all identified a majority of forestation area as occurring outside forest-dominated ecosystems. This suggests that the reported misplacement is unlikely to be driven solely by methodological choices. One possible reason for such substantial misplacement of forestation projects relative to forest ecosystems could be a divergence between operational definitions of "forest" used in practice and the ecological literature identifying where such forestation is feasible. For example, recent research highlights a tension between widely used definitions of 'forest' (such as that of the United Nations' Food and Agriculture Organization (FAO)) and an ecological perspective suggesting that reliance on vegetation structure (e.g. tree cover) may overlook key functional differences between ecosystems (Parr et al., 2024; Scogings, 2023). Under the FAO definition, ecosystems with over 10% tree cover can be classified as forest, meaning that savannas, and open woodlands may be included despite comprising shade-intolerant species and being maintained by disturbance regimes such as fire and herbivory (Charles-Dominique et al., 2018; Staver et al., 2011, 2021; Strömberg & Staver, 2022).

These definitions are not only semantic: the FAO definition is used for creating maps of forest coverage and as the standard for national forest inventories, which have subsequently been used to guide forestation efforts (Bourgoin et al., 2026; Chazdon et al., 2016; FAO, 2020; Johnson et al., 2023; Keenan et al., 2015; Qin et al., 2021). Similarly, many forest potential maps, which often guide the identification of areas for restoration or carbon sequestration, have been criticised for insufficiently accounting for disturbance regimes and other ecological constraints on closed-canopy forests (Bond et al., 2019; Fagan, 2020; Luedeling et al., 2019; Veldman et al., 2019). Ecologists emphasise that forestation in non-forest ecosystems is often inappropriate, as planted trees may be less resilient to disturbance and may alter existing ecosystems structure and function (Bond et al., 2019; Stevens & Bond, 2024; Veldman, Overbeck, et al., 2015a). The widespread use of forest definitions and suitability maps that may not fully capture ecological constraints may send normative signals that these areas are suitable for tree planting, even where they may be as or more suitable for open ecosystem states.

Our results are consistent with this interpretation. Misplacement is particularly pronounced in regions dominated by open ecosystems. This is most evident in Africa, where 80.4% of forestation area is classified as non-forest under all three frameworks, indicating clear agreement that these forestation sites are misplaced. Only a small fraction (2.9%) is consistently classified as forest across all frameworks, with limited partial agreement primarily between the predicted natural tree cover and ecoregion frameworks. Classified by ecoregion, two-fifths of forestation area across Africa was located in an open ecosystem, while the median predicted short vegetation was 0.45 (indicative of an open ecosystem). These results are consistent with a recent study that found the area pledged for reforestation in Africa outweighed the maximum degraded forest restoration area (Parr et al., 2024), but go further by proving that this is not only an issue at a theoretical (pledge) level but also in reality.

Misplaced forestation projects, particularly those located in open ecosystems, are potentially at threat from disturbances, particularly since a switch in fuel type from grasses to trees can cause more intense fires (Stevens & Bond, 2024). This can have ramifications for the sequestered carbon and thus the net zero targets that depend upon this, which is discussed further in the policy implication section (4.4.4). For the purposes of this section, however, it is noteworthy that definitional choices have real world impacts, affecting where forestation occurs and potentially contributing to ecological misplacement. Our results add evidence to the call for revised forest definitions (Sasaki & Putz, 2009; Sexton et al., 2016; Veldman, Overbeck, et al., 2015a; Zalles et al., 2024), which may go some way to addressing some of the issues identified in our results.

4.4.2. Why frameworks disagree

We expected that the extent of agreement between ecological frameworks would be limited, reflecting the different processes that they capture. Our results partially support this expectation. While a substantial proportion of forestation area is consistently classified across frameworks, agreement is less clear at a continental level. Most notably, agreement is highest in identifying areas where forestation does *not* occur within forest-dominated systems, whereas agreement in identifying forest-dominated placement is much lower. A substantial share of forestation area also occurs in regions where frameworks disagree, indicating uncertainty in ecological classification.

Our results are reflective of the differences in how the frameworks delineate forest-dominated systems. The ecoregion classification provides a broad, coarse-grained representation of ecosystem identity, while predicted natural tree cover captures variation in vegetation structure within these systems. In contrast, the biophysical forest potential framework applies more restrictive ecological constraints, resulting in a more conservative estimate of where closed-canopy forest is feasible.

This limitation is particularly evident in Oceania. Although more than four-fifths of forestation area falls within ecoregions classified as forest, very little of this area (1.6%) is consistently identified as forest-dominated across all frameworks. In particular, placement within areas identified as forest under predicted natural tree cover and biophysical forest potential is low (1.3% and 18.6%, respectively), indicating that the ecoregion framework is largely alone in classifying the forestation area as well placed. Forestation is concentrated in a small number of temperate forest ecoregions in eastern Australia and Tasmania. These regions are characterised by strong environmental gradients and mosaics of vegetation types, including mixtures of rainforest, eucalypt forest, and treeless or open systems occurring in close proximity (Kirkpatrick & DellaSala, 2011). Consequently, classification at the ecoregion level may overstate the extent to which forestation is placed within ecologically appropriate forest systems. The lack of placement of forestation sites within forest-dominated areas under the other frameworks suggests that many planting sites occur in parts of these ecoregions that are not suited to persistent closed-canopy forest, likely due to climatic constraints or disturbance regimes.

A similar pattern is observed in Europe. A large proportion of forestation area (66.9%, ~16 Mha) is classified as forest under only one framework, almost entirely driven by the ecoregion classification, while a further 29.7% is classified as non-forest under all frameworks. Very little forestation area is consistently classified as forest across all three frameworks. Forestation area is concentrated in a small number of forest-classified ecoregions, particularly across temperate mixed forest systems. These ecoregions encompass substantial internal heterogeneity, including mixtures of forest and non-forest systems and areas altered by long-term land use (Hengeveld et al., 2012; McGrath et al., 2015). The presence of forestation within treeless and mosaic ecoregions further indicates that projects are not consistently located in areas capable of supporting dense natural tree cover. This is likely as a result of the influence of long-term land use and forest management in Europe.

However, this pattern should be interpreted cautiously. In temperate Europe, including the UK, natural vegetation is often heterogeneous, comprising mosaics of closed-canopy woodland, scattered trees, shrubland, and open grassland. As such, classification of a site as lying outside a forest-dominated system does not necessarily suggest that tree planting or woodland restoration is ecologically inappropriate. Instead, our results highlight the importance of considering fine-scale ecological context when planning forestation.

Overall, the continental results indicate that estimates of forestation (mis)placement are sensitive to the ecological framework applied. While all three frameworks largely agreed on substantial misplacement at a global scale, the magnitude and spatial distribution thereof differ considerably. This has implications for policy design. Reliance on a single framework could risk mischaracterising the ecological appropriateness of forestation. Integrating multiple frameworks may provide a more robust basis for identifying areas where forestation is ecologically appropriate and likely to result in durable carbon storage.

4.4.3. Ecological and carbon implications

The widespread placement of forestation outside forest-dominated ecosystems has important implications for both ecological integrity and the durability of carbon storage. Across frameworks, a majority of global forestation area is consistently classified as non-forest, indicating that many projects are located in systems where closed-canopy forest is not the dominant or stable vegetation form. In such contexts, environmental constraints and disturbance regimes may limit the persistence of tree cover, increasing the likelihood that carbon stored through forestation is not maintained over time. This has direct implications for the durability of forestation as a climate mitigation strategy, particularly where long-term carbon storage is assumed (Anderegg et al., 2020; Brunner et al., 2024).

In open and disturbance-maintained ecosystems, including savannas and grasslands, tree planting may alter ecosystem structure and functioning. These systems are often maintained by fire and herbivory, and the introduction of trees can shift fuel structure, potentially increasing fire intensity and altering disturbance regimes (Bond, 2019; Bond et al., 2005; Hoffmann et al., 2012; Staver et al., 2011). Such changes may reduce the stability of newly established tree cover and increase the risk of carbon reversal through disturbance events. In addition, forestation in these systems has been associated with reduced water availability and changes in hydrological processes, particularly in semi-arid regions where water limitation constrains vegetation structure (Jackson et al., 2005).

Beyond carbon, forestation in non-forest ecosystems may have broader ecological consequences. Open ecosystems are often ancient, biodiverse systems with unique species assemblages and evolutionary histories, and planting blocks of closed-canopy forest can lead to declines in biodiversity and shifts in ecosystem function (Veldman, Overbeck, et al., 2015b). These changes may also affect the provision of ecosystem services, including those related to pastoral livelihoods and cultural uses of open landscapes (Skhosana et al., 2025). This is critical, since many poor households rely on ecosystem services and are particularly vulnerable to the loss thereof (Kumar & Yashiro, 2014). This reliance is particularly high in sub-Saharan Africa, which corresponds concerningly to where we found substantial misplacement.

These findings suggest that the ecological context in which forestation occurs is critical for determining both its environmental outcomes and its effectiveness as a climate mitigation strategy. Where forestation is located in areas consistently classified as non-forest across frameworks, the likelihood of achieving durable carbon storage may be reduced, and ecological trade-offs may be more pronounced. This underscores the importance of incorporating ecological suitability into the design and evaluation of forestation projects, particularly in the context of climate policies that rely on forestation as a primary mechanism for carbon dioxide removal (Brancalion & Holl, 2020; Di Sacco et al., 2021).

4.4.4. Policy implications

There has been a strong focus in the literature and policy on forestation as an important climate mitigation technique (Doelman et al., 2020; Griscom et al., 2017; Ménard et al., 2023; Reyer et al., 2009), leading to large numbers of projects already being implemented. Indeed, at present, forestation is the dominant form of carbon dioxide removal (S. Smith et al., 2024). The issues highlighted by our results – e.g. that forestation sites are located in non-forest ecosystems – raise concerns about the durability of stored carbon. This, in turn, puts the achievability of net zero at risk.

A third of entities with net zero targets explicitly cite that carbon credits will be used to reach the relevant target (Lang et al., 2024). Where such projects are located in areas consistently identified as non-forest across frameworks, the likelihood of carbon reversal may be higher, undermining the integrity of these credits. In order to safeguard the integrity of these targets, more stringent standards governing net zero targets, carbon offsets, and forestation are required (Axelsson et al., 2024). This applies both to voluntary frameworks and to emerging standard-setting efforts.

More broadly, these results align with concerns that current forestation policy often lacks sufficient consideration of local ecological context (Heilmayr et al., 2020). In many cases, forestation is promoted based on estimates of land availability or carbon sequestration potential without fully incorporating the ecological constraints that determine whether forest can be sustained (Aguirre-Gutiérrez et al., 2023; Luedeling et al., 2019; Veldman et al., 2019). This may contribute to the placement of projects in open or disturbance-maintained ecosystems, where carbon storage is less certain and negative ecological side-effects are possible.

Ou results show that in some ecological contexts, forestation may not be the appropriate choice. One implication is that forestation-specific targets may need to be reconsidered in favour of broader restoration targets instead (Tölgyesi et al., 2022; Vetter, 2020). Intact non-forest (and forest) ecosystems can act as an important carbon sink but are also valuable for their unique biodiversity and ecosystem service provision (Bai & Cotrufo, 2022; Dass et al., 2018; Feurdean et al., 2018). Efforts to restore ecosystems should continue to be supported, albeit with strong ecological safeguards.

4.4.5. Limitations and next steps

A global study invariably trades off the nuance and precision that comes with a finer scale. For instance, ecosystems can be heterogeneous at finer spatial scales, such that the forestation sites that we identify as falling outside of forest-dominated ecosystems could in fact be in ecologically appropriate areas. Riparian planting sites, for example, may be appropriate planting sites even where the broader ecosystem might not support forests (Fousseni et al., 2014; Jacobs et al., 2007). Further study would downscale the analysis to account for this.

With each of the secondary datasets used, there are inherent assumptions and limitations. The forestation dataset only included planting sites that had information available in English, publicly available, and machine-readable. Thus, the dataset may be misrepresentative of total forestation projects globally, since smaller-scale or more informal projects (i.e. those that are not reporting in English for the sake of international carbon markets) may be missed. Further, the database records planting sites over a 33-year period, which does not align with the static ecological frameworks that we used. While there is a temporal mismatch, we do not believe that there would have been a significant change in any of the three frameworks over the period in question. Since the locations were derived from self-reported project descriptions, there may be uncertainties pertaining to the planting site polygon boundaries and locations.

While we made efforts to remove projects that were not aimed at forestation using selected keywords, a number may still remain within the cleaned database. Linked to this, while we interpret misplacement as indicative of ecological misalignment, some forestation projects may be intentionally located in non-forest systems due to policy incentives, land availability, or socio-economic considerations. Distinguishing between intentional and unintended misplacement would be an important avenue for future work.

Moreover, our results may have been influenced by a lack of clarity in the literature and practice around the definitions of ‘afforestation’ and ‘reforestation’. While forest potential maps typically refer to terrestrial forestation, we also found several projects in the dataset that were aimed at planting native trees at an appropriate density in savannas. Given the importance of definitions, further work on articulating how afforestation and reforestation differ from restoration would be useful (Briske et al., 2024; Stanturf et al., 2026).

The spatial resolution of predicted natural land cover model is relatively coarse (0.25°) in comparison with the other frameworks. This may have led to a higher area of forestation being classified as either correctly or incorrectly placed than is the case in reality.

Last, the social implications of forestation locations are not taken into account in this study but would make an important contribution in future work (Enríquez-de-Salamanca, 2024; Fleischman et al., 2020). For example, local communities often rely on ecosystem services for basic needs (Fedele et al., 2021), but this analysis did not specifically explore how forestation sites with low placement within forest-dominated areas across the three frameworks could result in lowered ecosystem services to local residents. This is an important avenue for future research.

4.5. Conclusion

Across three complementary ecological frameworks, we find that forestation is predominantly located outside forest-dominated ecosystems, with only a small proportion of forestation area consistently classified as forest and a majority consistently classified as non-forest. Results differed across continents, with particularly high levels of misplacement in regions dominated by open ecosystems. These results indicate that ecological misalignment of forestation is an issue globally.

This has important ramifications for both the durability of carbon storage and broader ecological outcomes, particularly given the reliance on forestation within many net zero pathways. Where forestation occurs in systems that are not suited to forests, the likelihood of long-term carbon sequestration may be reduced. At the same time, forestation in non-forest ecosystems may alter ecosystem structure and functioning, with potential consequences for biodiversity and the provision of ecosystem services.

Our results suggest that greater scrutiny of where forestation projects are implemented is crucial. Ensuring that forestation is aligned with ecological context will be critical for safeguarding both ecosystem integrity and the effectiveness of forestation as a climate mitigation strategy.

Author contributions

CH: Conceptualisation, data curation, methodology, formal analysis, visualisation, writing – original draft, writing – review and editing; ZV: Conceptualisation, methodology, writing – review and editing; NS: Conceptualisation, methodology, writing – review and editing.

5. Comparing the benefits of native and novel ecosystems as urban nature-based solutions: a case study of Johannesburg, South Africa

> *I actually think Johannesburg represents the future. My version of what I think the world is going to become looks like Johannesburg.* – Neill Blomkamp
>
> *No second Johannesburg is needed upon the earth. One is enough.* – Alan Paton*, Cry the Beloved Country*

The final research paper in the thesis builds on the takeaway from the previous paper: that well-designed mitigation should take into account ecological considerations.A number of stakeholders in Johannesburg, including the municipal government itself, have identified tree planting as a promising mitigation technique (i.e. for carbon sequestration) for the city. However, as was discussed in the previous chapter, forestation in areas that are not naturally forest-dominated has the potential to raise risks relating to the ecosystem, communities, and the mitigation effort itself.

In this regard, Johannesburg is situated within the Highveld mesic grassland ecosystem, which is naturally grass-dominated with scattered woody species limited to rocky outcrops. Given that the urban area is predominately located on the grassy plains, this suggests that forestation may not be aligned with the natural ecosystem. Nonetheless, extensive afforestation has already occurred in some areas of the city, which has created a novel ecosystem within the urban area.

Thus, this paper considers the particular gap of cities in which there is a divergence between native and novel ecosystems, prompting questions around which should be used as the ecological reference state for nature-based-solutions design. We compare the native, novel and urban-industrial areas of the city on climate, biodiversity and human well-being benefits.

Ecological concerns are typically local and entangled with social concerns. As such, this chapter adds a social element to the context relevant to mitigation measures, using an integrated (nature-based solutions) approach. We then consider the distribution of these benefits across race and income.

This study offers an alternative approach to considering only the carbon potential of afforestation within Johannesburg.

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Comparing the benefits of native and novel ecosystems as urban nature-based solutions: a case study of Johannesburg, South Africa

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Urban nature-based solutions (NbS) must align with their socio-ecological context, yet there is limited evidence on what should serve as the ecological reference in cities with contrasting native and novel ecosystems. Using Johannesburg, South Africa, as a case study, we compare native, novel, and urban-industrial ecosystems in terms of climate, biodiversity, and well-being benefits, and consider their socio-economic distribution. We combine MODIS daytime land surface temperature (LST), a biodiversity intactness index, and national census datasets at a ward level to run regression models for cross-sectional analysis. Native- and novel-dominated wards are associated with lower LST and higher biodiversity than urban-industrial wards; native wards are also associated with higher use of provisioning ecosystem services. Higher income is correlated with cooler and more biodiverse wards. These associations suggest that protecting and expanding both the native grassland and the novel tree canopy could inform NbS design to support climate, biodiversity, and human well-being goals in Johannesburg and other similar cities.

Key words: ecosystem services, environmental justice, socio-ecological systems, biodiversity, climate regulation

5.1. Introduction

Climate change and biodiversity loss are already impacting infrastructure, livelihoods, and the provision of ecosystem services to urban residents (IPCC, 2023c; Sandifer et al., 2015). Urban nature-based solutions (NbS) offer a perceived 'win-win' solution by providing climate, biodiversity, and well-being benefits (Girardin et al., 2021). To ensure success, NbS project design and implementation must be in line with the relevant socio-ecological context, which comprises both ecosystem- and justice-based concerns (Kabisch et al., 2022; McPhearson et al., 2025). However, there is little clarity on what this means in practice.

To fill this gap, our research aim is to examine how native, novel, and urban-industrial ecosystems shape the benefits provided by NbS and whether there is a disparity in access to these benefits across race and income lines. To examine this, we ask: (1) do native-dominated wards in Johannesburg lower mean summer land surface temperatures, have higher biodiversity intactness, and provide food, water and fuel to more households than the novel or urban-industrial dominated wards? (2) Is a higher share of the concomitant benefits accruing to the White and wealthy residents of the city? We approach these questions through a city-wide comparison of the three ecosystems across the three dimensions of NbS: climate, biodiversity, and human well-being.

The literature is fragmentary, and often contradictory, about whether climate, biodiversity, and human well-being benefits in cities are associated with remnants of native ecosystems, or with novel ecosystems that comprise new assemblages of native and non-native species (Evers et al., 2018; Hobbs et al., 2006; Tartaglia & Aronson, 2024). Mostly, studies are focused on individual components (e.g. climate or biodiversity) or ecosystem type (e.g. novel or native), rather than comparing native and novel across multiple NbS benefits at a city scale. Of equal importance is an understanding of the potential justice ramifications of NbS. Provisioning ecosystem services provided by NbS are often relied on by lower-income households (Anderson & O'Farrell, 2012; C. Shackleton & Shackleton, 2004). Yet NbS may perpetuate or even create urban inequalities: higher tree cover, higher species diversity, and lower temperatures are often concentrated in areas with higher income and percentages of White residents (Benz & Burney, 2021; Leong et al., 2018; Venter et al., 2020). As such, a full understanding of the interplay between native and novel species thus also requires an examination of who is benefiting from each ecosystem, which remains undeveloped in the literature. In fact, a full examination of the literature makes it clear that two gaps remain, particularly in relation to urban NbS in the Global South: first, there is limited evidence of comparisons of native, novel, and urban-industrial across all three NbS benefits; and second, there is limited understanding how these benefits are distributed across race and income.

Johannesburg, South Africa, offers a useful case study, with a contrast between the remnant native grasslands and a novel ecosystem comprising an extensive planted urban forest, as well as clear racial and income-based divisions (Ballard & Hamann, 2021; Symes et al., 2017). In addition, the city is located in the Global South, which is underrepresented in the urban NbS literature (McPhearson et al., 2025), further strengthening its relevance as a case study.

We therefore compare wards dominated by native, novel, or urban-industrial ecosystems across three NbS dimensions: climate (land surface temperature), biodiversity (biodiversity intactness) and human well-being (provisioning ecosystem services), and assess whether race and income are associated with different levels of access to these NbS benefits. To guide the benefit dimensions that are examined, we draw on two concepts: (i) the NbS triad of climate, biodiversity, and human well-being benefits (Girardin et al., 2021); and (ii) Kowarik's four natures typology, which is used to underpin the ecosystem comparison and is re-conceptualised here as three ecosystems: 'native', 'novel', and 'urban-industrial' (Kowarik, 2011).

We predict that native-dominated wards will be cooler than urban-industrial wards; and have the highest biodiversity intactness, and percentages of households that make use of the ecosystem for food, water and fuel, as compared to both novel and urban-industrial. These predictions stem from the spatial configuration of Johannesburg, with native ecosystems often occurring in the less developed (and more ecologically intact) south of the city (Cilliers et al., 2013; Saaroni et al., 2018; Todes, 2012). Our hypothesis is that novel ecosystems will be linked to wards with higher levels of per capita income and White residents, given the city's history of unequal racial development (Mabin, 2014). Given the extensive literature on the cooling ability of urban trees, we predict that the novel ecosystems will be associated with the coolest temperatures overall (compared to both native and urban-industrial); and will have higher biodiversity intactness than the urban-industrial but lower than native (W. Li & Li, 2025; Nitoslawski & Duinker, 2016).

We predict that access to temperature regulation will be delineated by income and race, as has been found in other urban areas (Chakraborty et al., 2019; Grislain-Letrémy et al., 2025). We hypothesise that water, food and fuel provision will be primarily accessed by the urban poor, as a reliance on natural resources by the poor has already been observed in other African contexts (Cavendish, 2000; Cilliers et al., 2013; C. M. Shackleton & Shackleton, 2006). Given that a disproportionate number of the urban poor in Johannesburg are Black Africans (Ballard & Hamann, 2021), we predict that race will not limit access to water, fuel and food production. Lastly, given that the 'luxury effect' of higher species diversity correlated with higher income was disproved for avian species richness in Johannesburg (Howes & Reynolds, 2021),[*] we predict that income will not correlate with biodiversity intactness in Johannesburg.

5.2. Methods

We used secondary spatial and census datasets to compile ward-level metrics for climate, biodiversity, human well-being and socio-economic concerns. No primary field data were collected. Google Earth Engine was used for raster processing and zonal statistics (Gorelick et al., 2017); all statistical analyses, summary plot and map production were carried out in RStudio (Posit team, 2025). A summary of the variables used in this analysis is set out in **Table 3**.

[*] The 'luxury effect' is a pattern of higher biodiversity in the more affluent parts of urban areas, which has been observed in a number of cities worldwide (Leong et al., 2018)

5.2.1. Study area

Johannesburg is situated in the province of Gauteng, in the north-eastern part of South Africa (26°12' S 28°2' E) (**Figure 9**). It is located on an interior plateau - the Highveld - at altitudes of between 1081m and 1899m (Grobler et al., 2006). The city is situated in the Mesic Highveld Grassland bio-region, which is dominated by a grassy system with some wetlands and natural wooded ridges (C. Reynolds & Howes, 2023; Rutherford et al., 2006). The grassland biome has high levels of plant diversity (second in South Africa only to the fynbos biome) and is home to a wide variety of endemic animal species (SANBI, 2013). However, from early in the city's history, quick-growing (often alien) trees were planted for mining timber and colonial placemaking, predominantly in historically Whites-only suburbs (Bennett, 2017; Schäffler & Swilling, 2013; C. M. Shackleton & Gwedla, 2021). While there has been some attempt to expand the urban canopy to other areas of the city, it remains concentrated in the northern (former Whites-only) suburbs (Wright, 2022).

The city is governed at a local level by the City of Johannesburg Metropolitan Municipality. The city is split into 135 electoral wards, which is also the smallest spatial unit at which the Department of Statistics South Africa ('Stats SA') provides census data (Statistics South Africa, 2011). The national censuses, which have been undertaken in 1996, 2001, 2011, and 2022, collect social, economic, and demographic information on the South African population. Stats SA also conducts smaller-scale data collection, including community surveys that give municipalities information in the interim period between national censuses. The census wards served as the primary spatial unit for analysis, in line with prior work by the city government (City of Johannesburg, 2021a). The boundaries of the wards were sourced from the Municipal Demarcation Board (Municipal Demarcation Board, 2023).

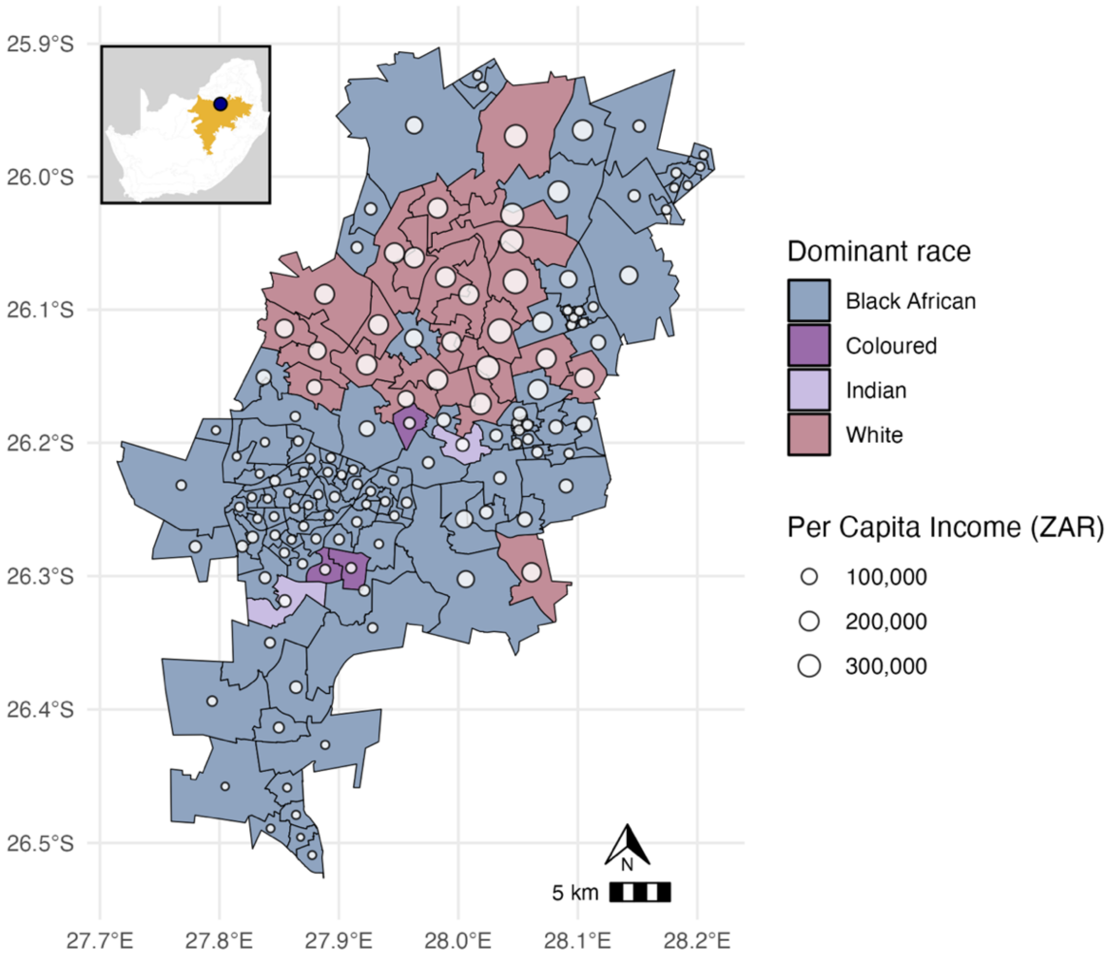

Figure 9: Income and race in Johannesburg. *Income and race in Johannesburg. The dominant race per ward is shown as the colour of each of Johannesburg's 135 census wards (Black African in blue; Coloured in purple; Indian in lavender; and White in pink), with the median per capita annual income (ZAR per person per year) in each ward as the size of the circle. The per capita income and dominant race in each ward were calculated from the 2011 national census (Statistics South Africa, 2011). Johannesburg's position in South Africa is shown as a blue dot in the inset. The extent of the Highveld ecoregion across South Africa is indicated in gold in the inset, derived from the South African Department of Water and Sanitation's ecoregion dataset (Department of Water and Sanitation, 2005).*

5.2.2. Urban ecosystem classification

To create a layer of ecosystem types we overlaid the South African National Land Cover (SANLC) 2020 footprint over Johannesburg's census wards. The SANLC dataset is generated by the South African Department of Forestry, Fisheries and the Environment, using Sentinel-2 satellite imagery at 20m resolution (Department of Forestry, Fisheries, and the Environment, 2020). It comprises 73 distinct land-cover classes. The proportional coverage of each land-cover class within each ward was extracted in Google Earth Engine. 56 of the 73 SANLC classes were found in the city's metropolitan boundaries.

We then recategorised the SANLC classes into three types of urban ecosystem: native, novel and urban-industrial (**Table S 9**), adapted from Kowarik (2011)'s four natures approach. The four natures approach is a generic typology categorising urban nature into four categories: pristine; agricultural; horticultural; and urban-industrial (Kowarik, 2011). We renamed Kowarik's pristine category to *native,* in response to the 'pristine myth' critique by Denevan (1992, 2011), which argues that the application of the word 'pristine' to the pre-colonial Americas is a misnomer, given that many landscapes have been managed – to greater or lesser extents – for centuries. Suggestions of pristine landscapes tend to be in line with colonial viewpoints of the landscape (Denevan, 1992, 2011). This applies equally well to the Highveld, which has been used for, e.g., cattle herding since at least the 15th Century (Sadr, 2020), and accordingly we prefer *native* to *pristine.* In this study, the native category refers to the extant indigenous ecosystems of the broader area in which Johannesburg is situated (viz. mesic Highveld grasslands with some forested ridges and wetlands) (C. Reynolds & Howes, 2023).

We collapsed Kowarik's agricultural and horticultural categories into one *novel* category, capturing ecosystems that contain a new assemblage of species compared to the native grassland biome, like gardens and the urban forest (Ahern, 2016; Kowarik, 2011). Both agriculture and horticulture fall under the umbrella of urban green space (like native), but the composition of the ecosystem created by both is entirely novel to the indigenous region. On this basis, we grouped the two as one category. In the context of the Johannesburg (and the wider province of Gauteng), agriculture and horticulture often co-occur in the same areas (Nino et al., 2020), and at 20m SANLC resolution they are not reliably distinguishable, making a combined category more appropriate for our analysis. Lastly, we use Kowarik's *urban-industrial* category, which refers to areas with high instances of grey infrastructure, like industrial sites or the city centre.

The percentage of each land-cover group per ward (which is set out in **Table S 10**) was aggregated from the relevant individual classes. Overall, Johannesburg's land cover is 27.17% native, 39.45% novel, and 33.39% urban-industrial. We then categorised each ward as being either native, novel or urban-industrial-dominated, depending on which land cover had the highest percentage in the ward (e.g. ward 1 would be categorised as urban-industrial as U-I covers 62.21% of the ward). We also ran models using the continuous proportion of each land-cover group per ward as predictors and found relatively similar results, justifying our use of categorical data for land-cover type.

Table 3: Dimensions of nature-based solutions (climate, biodiversity, human well-being/justice) and ecosystem type, with corresponding indicators, datasets, spatial scales, and roles in the regression models (RQ1, RQ2).

NbS Dimension	Indicator	Dataset	Scale	Role in analysis
Ecosystem type	Native, novel, and urban-industrial	South African National Land Cover (SANLC) 2020	20m	Predictor: ecosystem type in all NbS models
Climate regulation	Mean summer land surface temperature (°C), December 2020–February 2025	MODIS/061/MOD11A2	1km	Response: climate regulation benefit (RQ1; RQ2)
Biodiversity	Biodiversity intactness index (%)	Bii4africa project	1km	Response: biodiversity benefit (RQ1; RQ2)
Human well-being: provisioning ecosystem services	Poultry production: % households reporting poultry production	Statistics South Africa community survey, 2016	Ward level (% of households)	Response: individual provisioning ecosystem services (RQ1, RQ2)
	Livestock production: % households reporting livestock production	Statistics South Africa community survey, 2016	Ward level (% of households)	Response: individual provisioning ecosystem services (RQ1, RQ2)
	Vegetable production: % households reporting vegetable production	Statistics South Africa community survey, 2016	Ward level (% of households)	Response: individual provisioning ecosystem services (RQ1, RQ2)
	Cooking fuel: % households reporting use of wood as fuel for cooking	Statistics South Africa national census, 2011	Ward level (% of households)	Response: individual provisioning ecosystem services (RQ1, RQ2)
	Heating fuel: % households reporting use of wood as fuel for heating	Statistics South Africa national census, 2011	Ward level (% of households)	Response: individual provisioning ecosystem services (RQ1, RQ2)
	Water provision: % households reporting source of water as natural sources	Statistics South Africa national census, 2011	Ward level (% of households)	Response: individual provisioning ecosystem services (RQ1, RQ2)

NbS Dimension	Indicator	Dataset	Scale	Role in analysis
Human well-being: justice dimension	Dominant racial group per ward	Statistics South Africa national census, 2011	Ward level (population composition; dominant racial group used in models)	Predictor: justice dimension (race) in RQ2 models (effect of dominant racial group on NbS benefits)
	Per capita annual income (ZAR) per ward	Statistics South Africa national census, 2011	Ward level (per capita income, ZAR)	Predictor: justice dimension (income) in RQ2 models; also predictor of proportion of novel land cover

5.2.3. Mapping NbS benefits

For each NbS dimension we used a proxy indicator. For climate regulation, that was land surface temperature (LST). While LST is not always a good proxy for ambient air temperature and can amplify the cooling properties of trees (Chakraborty et al., 2022; Du et al., 2024; Venter et al., 2021), we were unable to source a near-surface temperature dataset at both a sufficiently fine scale and city-wide coverage for Johannesburg. We retrieved the LST data from the MODIS Land Surface Temperature (MODIS/061/MOD11A2) product through Google Earth Engine (Gorelick et al., 2017; Wan et al., 2021). The MODIS LST product provides an 8-day average LST at 1 km resolution. A drawback of the 1km resolution is its potential to miss variation at finer scales (Kim et al., 2025), and thus our LST values are best interpreted as ward-level surface temperature conditions rather than street-level air temperatures. However, wards in Johannesburg are on average 12.41 km^2, and thus contain on average 12 MODIS pixels, which means that the 1km resolution is appropriate for our ward-scale analysis. We filtered the MODIS dataset to the daytime LST band to calculate the mean LST for the summer months (December through February) of each year from December 2020 through February 2025. This was used to extract a mean LST over the 5 years for each of Johannesburg's 135 wards.

To generate an understanding of indigenous biodiversity, we used the biodiversity intactness index (BII), a widely used indicator of biodiversity loss (Liu et al., 2025). The BII gives an indication of the average abundance of a wide variety of organisms in a given area, relative to a reference state (Scholes & Biggs, 2005). It is often used to assess the health or condition of an ecosystem. The BII is applicable across a wide range of spatial and temporal scales, which makes it a useful indicator for policy and planning (Clements et al., 2026). We used data from the bii4africa project (https://bii4africa.org/), which is an expert-solicited dataset of biodiversity intactness specifically for Africa at a scale of 1km, based on the knowledge of 200 experts in African fauna and flora (Clements et al., 2024, 2026). In Google Earth Engine, the bii4africa data was overlaid over Johannesburg to calculate the mean biodiversity intactness per ward. While we had considered adding species richness as a further biodiversity metric, the available data were via citizen science platforms and we found significant clustering in certain areas of the city, suggesting collection bias (Garzon Lopez & Savickytė, 2023).

We were interested in human well-being from two different perspectives. First, we were concerned with the provisioning benefits that are provided by the different types of ecosystems within Johannesburg. For this question, we considered fuel, food and water provision, both for their value in fulfilling basic human needs and for reasons of data availability. Second, we were interested in the justice component of NbS, thus investigating the distribution of the NbS benefits across race and income. All data relating to human well-being were secondary and were extracted from the 2011 national census and 2016 community survey databases provided by Stats SA (data available at https://www.statssa.gov.za/), which, as explained above, is a collection of economic, social and demographic statistics about the South African population. The census provides consistent data across the city, assisting in comparisons of NbS benefits across wards. The revised 2011 census (which realigned the original 2011 data to be in line with revised 2016 municipal boundaries) was used for all categories (fuel provision; water provision; race; income) (Statistics South Africa, 2011), except for the food provision category, which included data on household involvement in agricultural activities from the 2016 community survey (Statistics South Africa, 2016). We did not use the 2022 census as the ward-level data had not yet been released at the time of writing.

For each of the categories, the raw number of households that use that service was converted into percentages of total households in the ward, as a proxy to represent the level of provisioning ecosystem service use in each ward based on previous approaches in the literature (see **Figure S 4** for the spatial spread of the use of each provisioning service) (Hamann et al., 2015; C. Reynolds et al., 2020). For food provision, three of the 2016 household agricultural datasets were used: livestock, poultry, and vegetable production. Fuel provision was represented by two metrics: the type of fuel each household used as its main source for cooking or heating, which, given the design of the census questions, were not combined into one fuel metric but were evaluated separately (Hamann et al., 2015). Respondents to both census questionnaires (cooking and heating) could choose one fuel type from a list of 10, comprising, amongst others, electricity, gas, paraffin, coal, candles, animal dung, solar, and wood. The provision of fresh water was measured through data relating to a household's main source of water for household use. A range of sources were presented to the questionnaire respondents: regional or local water schemes (i.e., water services provided by the municipality), boreholes, springs, rainwater tanks, dams, rivers or streams, and water vendors. We used springs, and rivers or streams, as proxies for water provision, grouping the two as one overarching category; i.e., water retrieved from natural sources (C. Reynolds et al., 2020).

Across all provisioning ecosystem services (livestock, poultry, vegetable production, water provision, and use of wood for cooking and heating), there were a number of pairs that had significant and moderate to strong correlations (**Table S 11)** i.e., both fuel provision categories; water and wood for cooking; as well as the three food production services (livestock, vegetable, poultry). There were also a number that had significant but weak positive correlations. With this in mind**,** we considered each of the provisioning services separately but also combined them to find a single principal component of provisioning ecosystem service per ward following C. Reynolds et al. (2020). We undertook principal component analysis (PCA) on percentages of each of the six provisioning variables (livestock, poultry, vegetable production, water provision, wood used for cooking fuel, and wood used for heating fuel) for all 135 wards, using the *prcomp* function from the *stats* package in R. All variables were centred and scaled prior to PCA so that each provisioning variable contributed equally in terms of variance. However, we did not weight each provisioning service equally; instead, contributions were dependent on the loadings from the PCA. The first principal component (PC1) explained 45% of the total variance and thus we interpreted it as an index of overall provisioning ecosystem services. Higher scores on the index indicate a higher combined use of ecosystem services. We linearly rescaled PC1 scores to be between 0 and 1 for all wards.

Loadings on PC1 were positive for all six provisioning variables, with poultry production (0.51), vegetable production (0.48), and livestock production (0.45) contributing most strongly, followed by wood used for cooking and heating fuel (0.37 and 0.36 respectively), and water provision contributing most weakly (0.20). This pattern indicates a shared underlying gradient of direct provisioning ecosystem service use, without dominance by a single indicator. We therefore interpreted PC1 as an index of overall provisioning ecosystem service use.

For race, the realigned 2011 census data reports five categories for each ward: 'Black Africa', 'White', 'Coloured', 'Indian' and 'Other'. Using population numbers for each ward, we found the percentage racial category composition per ward and categorised each ward per its dominant racial category. While we considered categorising wards by race where that race was over a certain threshold, we found this skewed the categorisation heavily in favour of Black African and Mixed (i.e. those where there was no racial group over the threshold) wards. Instead, we opted for using the race with the highest percentage, finding 106 Black African wards as compared to 24 White wards, 2 Indian, and 3 Coloured. Since the Indian and Coloured wards were comparatively small, we predominantly focused on the White- and Black African-dominated wards. The dominant race categories were used as predictors in race-NbS models.

It is important to note that some readers may take issue with the use of the racial categorisations like ‘White’ and ‘Black African’. However, these categories reflect the history and lived reality of many South Africans, as the country continues to grapple with the insidious legacies of Apartheid. The racial categories are in informal and formal use within the country (Madlalate, 2017), and this article follows that convention. This does not suggest that the use of these categories should be accepted without further pause; here it is merely for the sake of continuity with the literature and country context.

Annual household income from the national census is split into twelve income brackets ranging from R0 to ~ R2.46m. Mean interval values for each income bracket were used to estimate total income per ward, which was then divided by the number of residents to calculate per capita income in each ward.

5.2.4. Statistical and spatial analyses

All statistical analyses and spatial depictions were carried out in RStudio, using the tidyverse, sf, performance, and stats packages (Lüdecke et al., 2021; Pebesma, 2018; Posit team, 2025; Wickham et al., 2019). We fitted a series of regression models to explore the relationship between the NbS benefits and (i) land-cover type; and (ii) race and income, so as to answer our research questions on the differences between ecosystem types in providing NbS benefits, and how these are distributed across race and income. Separate models were fitted for each response variable (like temperature or biodiversity intactness), as the predictors and scales of variation differed. Normal distribution was tested through QQ plots and Shapiro-Wilk tests. Where the data were normally distributed (like land surface temperature), we fitted linear regression models, while for non-normal distribution data (like poultry production), we used generalized linear models, selecting the error distribution (Gamma, Poisson or Inverse Gaussian) based on model fit criteria (AIC and residual diagnostics). For Gamma generalized linear models, R's default inverse link was used. Accordingly, coefficient signs are presented on the link scale, while ecological interpretation is based on the response scale. Each model was tested for linearity, homoscedasticity, influential outliers, the normality of the residuals, as well as spatial autocorrelation. We used Moran's I to test for spatial autocorrelation. Where there was evidence of significant autocorrelation ($p < 0.05$), we tested our models against spatial error models, using the errorsarlm function in the spatialreg package (Bivand et al., 2021). Although spatial error models often reduced AIC, coefficient signs and magnitudes remained similar and did not alter the substantive interpretation of results, justifying our choice of linear and generalized linear models (see **Table S 12** for spatial model outputs).

Land-cover type (encompassing the dominant land cover per ward) and race (the dominant race per ward) were categorical variables, but all other variables were continuous. Where one ward had a value of 0 for certain variables (i.e. water collection), this sole ward was excluded from data for GLMs with log-link and Gamma/Inverse errors, as these distributions require strictly positive responses. Less importance was placed on statistical significance in line with developments in the statistics literature (Amrhein et al., 2019; Muff et al., 2022; Nature Editors, 2019). All regression coefficients are presented with their standard errors (± SE).

5.3. Results

This section is organised around our two main research questions. We first present the results of the climate, biodiversity, and human well-being benefits associated with native, novel, and urban-industrial ecosystems at ward level. Second, we present the results of how these NbS benefits are distributed across race and income. We present model estimates from the relevant regression models (see Methods, Statistical and spatial analyses) and illustrate patterns using maps and summary plots. Full model coefficients, standard errors and p-values for all model outcomes are set out in **Table 4** with additional results in **Table S 13**; below we summarise the main effect sizes and patterns. Following Muff et al. (2022), in the text we describe evidence as: no or little ($p > 0.1$); weak ($0.05 < p \leq 0.1$); moderate ($0.01 < p \leq 0.05$); strong ($0.001 < p \leq 0.01$); and very strong ($p \leq 0.001$) (Muff et al., 2022).

5.3.1. Ecosystem type and NbS benefits

To test our first research question, we modelled the dominant ecosystem type in each ward against land surface temperature, biodiversity intactness, and ecosystem services (wood provision for fuel and cooking; water provision; vegetable, poultry and livestock production; and a composite of all the ecosystem services).

5.3.1.1. Clear clustering of native, novel and urban-industrial areas

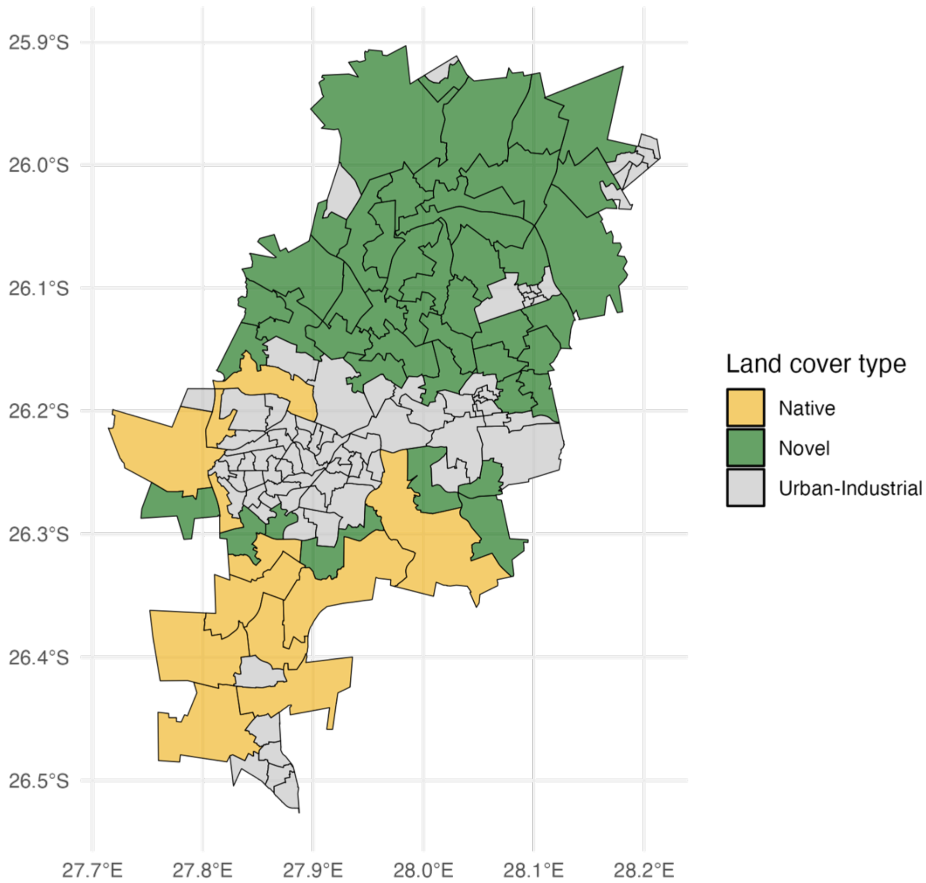

Figure 10: Dominant land cover by ward in Johannesburg. *Each ward was classed by the highest percentage land-cover type, represented in this figure by grey for urban-industrial, gold for native, and dark green for novel. The urban-industrial wards cover the original central business district and the township Soweto, while the novel wards fall in the 'northern suburbs', which were designated Whites-only during Apartheid.*

As a preliminary step, we were interested in the distribution of the ecosystems across the wards, mapping them spatially across the city. We found that the wards dominated by native, novel, or urban-industrial land cover were largely clustered together in three major groups in Johannesburg (**Figure 10**). The novel ecosystem-dominated wards are largely in the north of the city, with a middle band of urban-industrial, and native ecosystems on the southern periphery of the city.

5.3.1.2. Native and novel ecosystems offer climate and biodiversity benefits

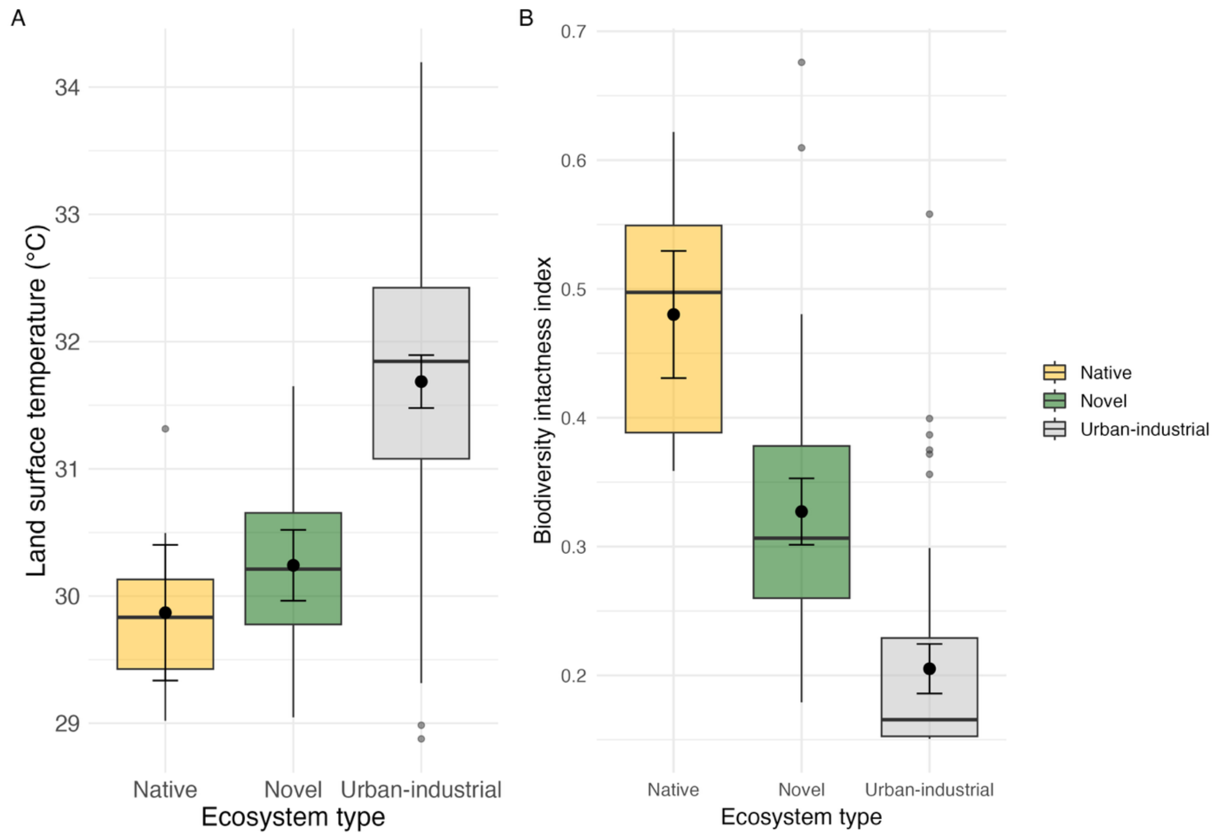

Figure 11: Land surface temperature in °C (A) and biodiversity intactness (%) (B) across urban-industrial (grey), native (gold) and novel-dominated (green) wards in Johannesburg. *Black points and error bars represent means and ± 95% confidence intervals from regression models. Novel and native ecosystems were associated with substantially lower temperatures and higher biodiversity intactness, as compared with urban-industrial wards.*

Novel- and native-dominated wards were associated with substantially lower land surface temperatures than urban-industrial wards (**Figure 11**). Native-dominated wards were on average cooler (29.9ºC) than both novel (30.2ºC) and urban-industrial (31.7ºC). In the regression models, native- and novel-dominated wards had markedly lower LST than urban-industrial wards (**Table 4**). Compared to urban-industrial wards, native wards were on average 1.82ºC (± 0.29) cooler and novel wards 1.44ºC (± 0.18) cooler. Panel A of **Figure 12** shows the clustering of the higher temperature wards in the south-west of the city, which is largely associated with urban-industrial land cover.

Native- and novel-dominated wards also offered biodiversity benefits compared to urban-industrial wards (**Figure 11**). Native-dominated wards had the highest biodiversity intactness, both on average (48% compared to 32.7% for novel and 20.5% for urban-industrial) and in the regression models (**Table 4**). The low biodiversity intactness in the urban-industrial wards in the south-west of Johannesburg, and higher intactness in native wards in the southern periphery and novel wards in the northern suburbs, is clear in Panel B of **Figure 12**.

Table 4: Regression model coefficients for the effects of land cover and demographic data on NbS benefits. *Estimates are presented along with standard errors and p-values. For brevity, only models with strong or very strong evidence (p < 0.01) are presented here. Full regression results are provided in* ***Table S 13****. Income is measured in ZAR per capita per year; coefficients for income appear very small in this table because they reflect the effect per ZAR 1. In the text, we interpret these effects per ZAR 100 000 increase in income for readability. Novel cover is expressed as the proportion of ward area. The provisioning services index is rescaled 0-1. See Methods for further details on model specification and variable scaling.*

Response	Predictor	Estimate	Std. Error	P-value
Land surface temperature	Native	-1.82	0.29	4.52E-09
	Novel	-1.44	0.18	1.64E-13
	Income	-5.83E-06	9.63E-07	1.37E-08
Biodiversity intactness	Native	0.27	0.03	2.00E-16
	Novel	0.12	0.02	8.16E-12
	Income †	-3.84E-06	1.28E-06	3.19E-03
Provisioning services	Native	2.72	0.45	1.33E-08
	Novel	0.88	0.27	1.66E-03
Poultry production	Native †	-0.37	0.11	8.03E-04
Livestock production	Novel †	-0.33	0.10	1.45E-03
	Income †	-1.42E-06	3.62E-07	1.45E-04
Vegetable production	Native †	-0.31	0.06	1.54E-07
	Novel †	-0.16	0.06	7.26E-03
	Income	1.24E-06	4.75E-07	8.77E-03
Cooking fuel	Native †	-3.18	1.16	6.90E-03
Heating fuel	Native †	-0.79	0.19	7.64E-05
	Novel	-0.59	0.20	3.13E-03
Novel cover	Income	1.63E-04	1.20E-05	2.00E-16
	White	0.85	0.12	1.37E-12

5.3.1.3. Native and novel are positively linked to provisioning ecosystem services

We ran regression models on a composite provisioning-ecosystem-services metric and found very strong evidence of a positive association with native-dominated wards (**Table 4**). The

† Gamma generalized linear model (inverse link). Coefficients are presented on the link scale and should not be interpreted directly as increases or decreases in the response variable.

highest values of the composite provisioning index occurred in wards on the periphery of Johannesburg, particularly in the north-west, east, and south (Panel C of **Figure 12**).

Further, on an individual level, this pattern also held (**Table 4**). In addition to a spatial clustering of the vegetable production in the southern periphery of the city (Panel D of **Figure 12**), the models showed very strong evidence that native-dominated wards had a higher expected percentage of households that engaged in vegetable production, as compared to the urban-industrial-dominated wards. For novel-dominated wards, too, there was strong evidence of higher expected vegetable production as compared to urban-industrial.

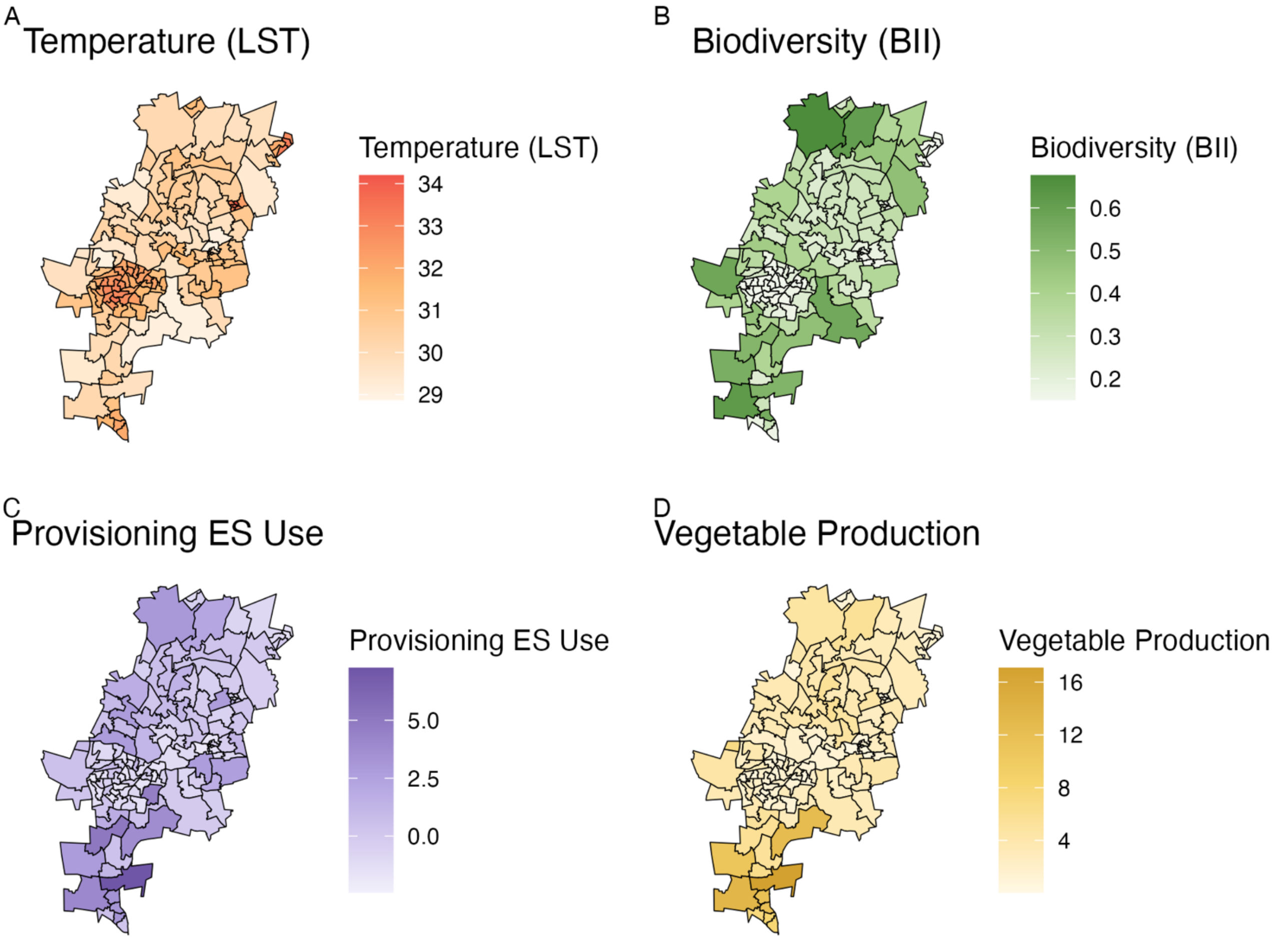

Figure 12: Spatial spread of temperature, biodiversity, provisioning ecosystem services, and vegetable production in Johannesburg. *The panels show the mean summer land surface temperature in °C (A); mean biodiversity intactness (%) (B); the use of provisioning ecosystem services, scaled between 0-1 (C); and the percentage of households that engage in vegetable production (D), per ward. Each panel uses a sequential colour scale in which darker colours indicate higher values, with red tones for temperature, green for biodiversity, violet for provisioning ecosystem service use, and yellow for vegetable production.*

Further, there was very strong evidence that native-dominated wards had a higher expected percentage of households that use wood as the main source of fuel for heating and strong evidence of the same for cooking (**Table 4**). We also found strong evidence of a higher expected percentage of households using wood as a heating fuel source in novel-dominated wards. We found no evidence of linkages between water provision and any of the variables.

5.3.2. Distribution of NbS benefits across race and income

To test our second research question, we modelled race and income against the ecosystem type (native, novel, and urban-industrial) and the NbS benefits (climate, biodiversity, and human well-being).

5.3.2.1. Lower income limits access to novel ecosystems and climate benefits, with positive associations with biodiversity intactness and livestock production

Average per capita income per ward ranged from ~ R6 300 to ~ R391 000 and was highest on average in the northern, White-dominated wards (~ R238 000). Our models showed very strong evidence that novel-dominated wards had a positive relationship with income, such that an increase of R100 000 in per capita income was linked to a 16.27 (± 1.2) percentage-point increase in the proportion of novel land cover (**Table 4**). We found very strong evidence that for every R100 000 increase in per capita income, LST decreased by 0.58ºC (± 0.1). In addition, we found strong statistical evidence of a small positive association with vegetable production.

There was strong evidence that income had a positive relationship with biodiversity intactness. We found very strong evidence that income was positively linked to livestock production. However, the percentage of households that undertake livestock production in any given ward is low (<3.5%).

5.3.2.2. White-dominated wards linked to novel ecosystems but otherwise race is not a clear predictor of climate, biodiversity, and human-well benefits

While **Figure 13** indicates that Black Africans tend to live in the hottest wards, the regression models found no link between the dominant racial group in a ward and mean summer land surface temperature once income was accounted for. White-dominated wards had on average the highest percentage of novel land cover (57.1% of the ward), the coolest temperature (30.19ºC), the highest biodiversity intactness of the race-dominated ward categories (30%), as well as the highest number of households that engaged in vegetable production (3.8%) and that used wood as their main source of heating fuel (2.4%). However, after controlling for income the racial composition of a ward did not emerge as a strong predictor in any of our regression models.

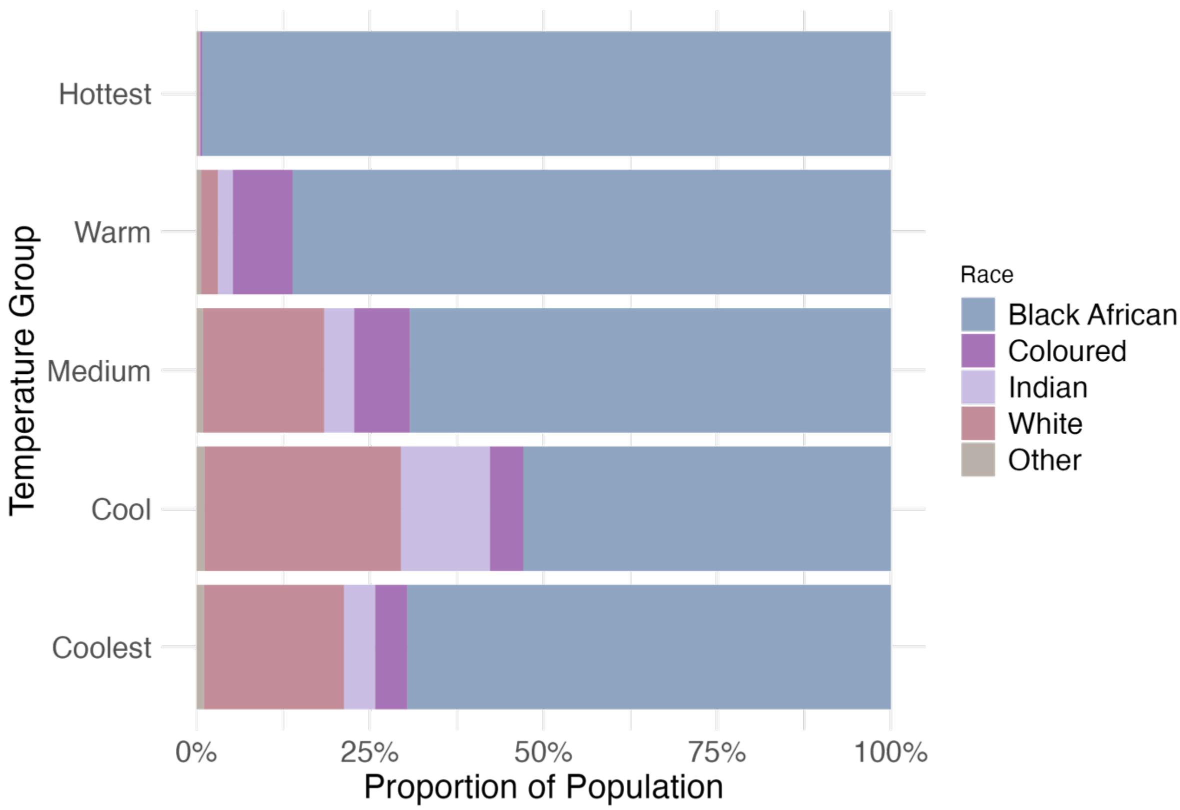

Figure 13: Race and temperature in Johannesburg. *The proportion of each racial group living in wards from the coolest to the hottest temperature quintile (Black African in blue; Coloured in purple; Indian in lavender; White in pink; and Other in grey). To divide wards into temperature groups, mean summer land surface temperature values were split into quintiles (coolest to hottest). The hottest wards are predominantly those with high percentages of Black African residents. These patterns are*

*descriptive. The regression models (**Table 4**) indicate that these temperature differences are largely explained by income rather than race per se.*

Income did prove to be a strong predictor, which suggests that the results of **Figure 13** and the averages set out above are more closely linked to income than race. In other words, once income is accounted for, race itself explains little additional variation in NbS benefits. The notable exception is very strong evidence of a positive link between White-dominated wards and novel land cover (**Table 4**), as is clear in the tree cover distribution shown in **Figure 14**, which reflects the racialised history of tree planting in Johannesburg.

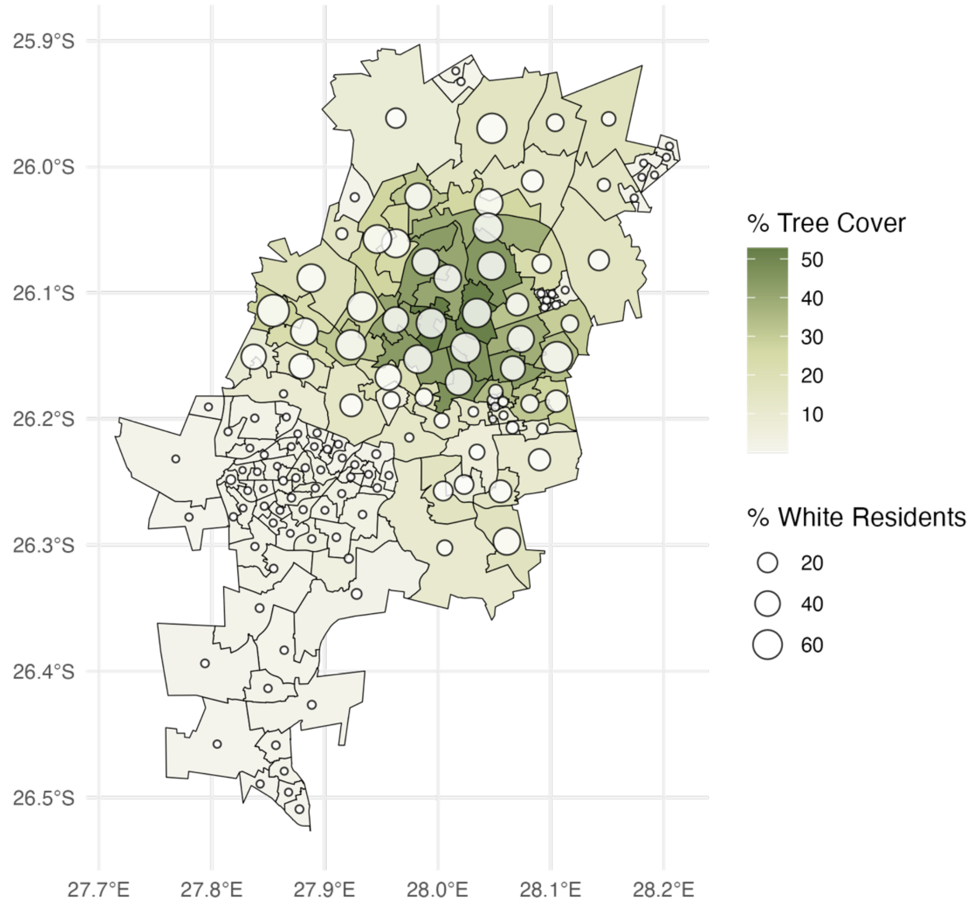

***Figure 14: Johannesburg's tree cover and percentage of White residents by ward.** Percentage tree cover is depicted on a gradient from 0 (light cream) through approximately 55% (dark olive green), with the percentage of White residents represented by circle size (smallest being the lowest percentage). Tree cover calculated as the percentage of pixels per ward that are classed as class 10 (trees) of the European Space Agency's world cover dataset, which provides land-cover classifications at 10m resolution (Zanaga et al., 2021). The percentage of White residents in each ward comes from the 2011 national census (Statistics South Africa, 2011). The urban forest of the northern suburbs is clearly contrasted against the more industrial and grassy southern suburbs.*

In the aggregate, the results suggest that native ecosystems offer competitive benefits as compared to the novel ecosystems, with both offering a suite of benefits. In the following discussion, we explore the implications for urban policy in Johannesburg.

5.4. Discussion

In this study, we examined how native and novel ecosystems link to NbS benefits. Specifically, we asked if native ecosystems in Johannesburg lower mean summer land surface temperatures, have higher biodiversity intactness, and provide food, water, and fuel to more households than the novel or urban-industrial areas of the city. In tandem, we also examined whether a higher share of these nature-based benefits was accruing disproportionately to the White-dominated and wealthier wards. In the following paragraphs, we first discuss the ecological implications of our results, followed by a consideration of justice. We follow this with policy takeaways, and limitations and future research directions.

5.4.1. Ecological implications

Given extensive research on the cooling effects of trees in urban areas (Loughner et al., 2012; Ribeiro et al., 2023; Sharmin et al., 2023; Tan et al., 2016; Wang & Akbari, 2016; Wu et al., 2025; Ziter et al., 2019), we theorised that novel ecosystems in Johannesburg would have the coolest mean summer land surface temperature (LST) compared to the other ecosystems. However, we found that while novel-dominated wards had lower land surface temperature (on average 1.44ºC cooler) than urban-industrial wards, there was an even greater difference (on average 1.82ºC cooler) for native-dominated wards. One possible reason for this is that native ecosystems in Johannesburg are primarily located in the southern periphery of the city. Accordingly, lower mean summer LST in native-dominated wards may be partly explained by the urban heat island (UHI) effect, where higher building density has been linked to higher UHI intensity (Y. Li et al., 2020). As can be seen in our land-cover maps, the novel and urban-industrial ecosystems are more centrally located and coincide with the urban core (for example, Braamfontein and surrounding suburbs for the urban-industrial, and Sandton and surrounds for the novel). These centrally located areas are hotter than the city's southern periphery in our LST data, consistent with a UHI pattern (Y. Li et al., 2020; Peng et al., 2012; Stewart & Oke, 2012). Further, since we did not include other drivers like elevation or local climate gradients, we stress that this result is merely an association, not causation. Regardless, the implication of our results is that native grassland ecosystems can also act as effective NbS for cooling, alongside the treed novel ecosystems in the city. The temperature difference between the urban-industrial and both the native and novel ecosystems suggests that, from an NbS design perspective, the City of Johannesburg should prioritise the protection and restoration of native grassland areas, as well as the avoidance of further conversion of the native or novel ecosystems to urban-industrial cover. At a neighbourhood scale, temperature reductions of around 1-2ºC have been linked in other urban contexts to reductions in heat stress and heat-related mortality risk (Arbuthnott et al., 2020; Murage et al., 2020), suggesting that the magnitude of the LST differences we observe is relevant for Johannesburg's climate resilience and human well-being.

Biodiversity intactness was also linked to the native-dominated wards in Johannesburg. The higher levels of biodiversity in the native wards are likely explained by the location of the native ecosystems in less developed areas of the city, which has meant that less disturbance of the original ecosystem has occurred. This is worth flagging given that further development in Johannesburg, if not carefully managed, could undermine biodiversity intactness, in line with a concern that has been raised about a potential trend of urbanisation and biodiversity loss on the African continent (Ayeni et al., 2023). This is particularly relevant to the native grasslands of Johannesburg, as grassy ecosystems and their associated, often endemic, biodiversity are declining globally (Bardgett et al., 2021; Parr et al., 2014; Stevens et al., 2022). Ensuring that pockets of ecosystems are protected in urban areas is critical for maintaining biodiversity (Beninde et al., 2015), which can have beneficial impacts on human health and well-being

(Carrus et al., 2015; Le Provost et al., 2023; MacKinnon et al., 2019; Wood et al., 2018). Careful development that incorporates climate, biodiversity and human well-being benefits is thus critical in Johannesburg going forward.

We found that Johannesburg's native-dominated wards are associated with higher composite provisioning ecosystem service use than novel and urban-industrial wards in the city, and that there was evidence to support the contention that native wards had more households accessing individual food-, and fuel-based ecosystem services. However, there was no evidence to support a relationship between water provision and native-dominated wards. The almost complete lack of a relationship between water sources and land cover can be explained by the fact that most of the water supply in the city is provided by a state-owned water scheme (City of Johannesburg, 2018). South Africa has a globally-recognised progressive approach to the provision of water: the Constitution grants everyone the right to "sufficient" water (Constitution of the Republic of South Africa, 1996), which has been actioned by the government through the provision of water services (with a base amount provided for free) to all members of society (Humby & Grandbois, 2011). Johannesburg has been one of the best performing municipalities in ensuring access to water services with 98.8% of households having access in the city (City of Johannesburg, 2021b; Humby & Grandbois, 2011). Accordingly, less water may be harvested from natural sources like rivers or springs.

While NbS in cities is well-established in the literature, there has not yet been consensus on whether urban NbS interventions should comprise native or novel assemblages of species, or even whether it should be a mix. In addition, the cooling ability of trees has been extensively covered in the literature. Our results showed that both native and novel ecosystems in Johannesburg offered climate, biodiversity, and human well-being benefits, indicating that the choice between native and novel is context dependent. However, the choice is unlikely to be a binary, and a mixture of native and novel species may be effective in certain contexts.

5.4.2. Justice implications

We expected White-dominated and wealthy wards to benefit disproportionately from the benefits of NbS in Johannesburg. While White-dominated wards were positively linked to novel ecosystems, we found no significant relationships between any of the NbS benefits and race after controlling for income. Our analysis treats race and income as separate predictors, but in Johannesburg these are deeply intertwined, with clear spatial patterns that reflect the enduring legacy of Apartheid spatial planning. Poorer Black African households are disproportionally situated in urban-industrial-dominated wards in the city, which are hotter and have lower levels of tree canopy. While our dominant racial categorisation at a ward level cannot capture the nuance within the ward, it does highlight the importance of intersectional approaches that consider race and income together when designing NbS interventions.

We found that income is positively linked to novel wards, biodiversity intactness, livestock production, and vegetable production, but negatively linked to temperature. The biodiversity-income result supports the 'luxury effect' trend found in cities globally and elsewhere in South Africa (Anderson et al., 2020; Leong et al., 2018) but contrasts with the findings of a recent paper, which found no evidence of the luxury effect using bird richness and diversity in Johannesburg's green spaces (Howes & Reynolds, 2021). This contrast may be explained by our use of a broader biodiversity metric and wider spatial scale (city-wide), compared to their focus on avian diversity within public green spaces. Our livestock-income result should be approached with caution given that the absolute production is extremely low in most wards.

The well-documented 'green divide' between the wooded former Whites-only areas and the rest of the city, a legacy of Apartheid (Abutaleb et al., 2021; Schäffler & Swilling, 2013; van Staden & Stoffberg, 2021), was confirmed by our results, with novel ecosystems being clearly linked to White-dominated wards. The link between White residents and trees is not unique to Johannesburg, however, and our results aligned with literature linking areas with higher percentages of white residents in North American and other South African cities to higher tree cover (W. Li & Li, 2025; Locke et al., 2021; McDonald et al., 2021; Venter et al., 2020). However, the benefits are not racially patterned after accounting for income. Overall, we found that income, more than race, structures access to NbS benefits in Johannesburg, with exception of legacies of racialised tree planting, which leave White-dominated wards as being closely linked to novel-dominated wards with their high levels of tree cover.

We found that income had a negative relationship with land surface temperature, suggesting that poorer households are likelier to live in hotter wards in Johannesburg. This aligns with the UHI literature that finds a correlation between income and heat (Chakraborty et al., 2019; Grislain-Letrémy et al., 2025; Raj et al., 2025). While this raises clear justice concerns, it is critical that this result is understood against the background of a strong normative discourse that 'trees are good' (Roman et al., 2021). Our results complicate this narrative by showing that, in the context of Johannesburg, grassy native ecosystems can also provide climate, biodiversity and human well-being benefits, complementing the benefits accruing from the urban canopy.

In practice and in theory, NbS are often positioned as a 'triple win'. This framing has been contested on the basis that it may obscure the potential inequities and injustices that can arise out of NbS projects (Bauer, 2023). Our results show that higher per-capita income in Johannesburg is correlated with lower temperatures, suggesting that the distributive effects of NbS are critical to consider in the design process.

5.4.3. Implications for NbS design and urban planning in Johannesburg

Our results support a recent assertion by McPhearson et al. (2025): "local ecological contextual knowledge is critical for the efficacy of nature-based solutions." In Johannesburg, we find that both native and novel ecosystems are associated with cooler temperatures and higher biodiversity intactness compared to urban-industrial areas, with native-dominated wards also showing higher composite provisioning ecosystem service use.

These results highlight three main implications for urban planning, ecosystem management, and climate justice in Johannesburg. First, both the native- and novel-dominated wards were on average cooler than the urban-industrial, suggesting that these ecosystems provide a significant climate regulation service that could be harnessed in pursuit of Johannesburg's climate adaptation goals. Both the urban canopy and the extant natural grasslands contribute to cooling and thus should be protected and expanded through spatial planning and green infrastructure strategies. Second, expansion of NbS in Johannesburg should be prioritised in the lowest-income wards (which overlap significantly with the urban-industrial), given the clear relationship between income and temperature that we found. Targeting hotter, low-income wards for grassland restoration and tree planting would align climate adaptation with climate justice goals and would be consistent with the City of Johannesburg's own commitment to addressing social justice as part of its climate strategies (City of Johannesburg, 2021a). Third, our results indicate that protection and restoration of the native grasslands within the metropolitan area is crucial for maintaining biodiversity. The current focus of the City of Johannesburg is tree planting (City of Johannesburg, 2021a), but our results suggest that the local government should explicitly recognise grasslands as a critical component of Johannesburg's green infrastructure and biodiversity planning.

The protection of grasslands within Johannesburg does not need to come at the expense of the urban canopy. There are several mechanisms for urban planners to ensure both trees and grasslands are present in the city. An example would be a recognition that, in some contexts within the city, fewer trees may be more appropriate. Road verges, highway medians, and post-mining landscapes (such as those in the south of the city) are particularly suitable candidates for expanding native grassland cover. Further, urban residents in South African cities often associate urban trees with higher rates of crime (S. Shackleton et al., 2015), suggesting that grasslands may be appropriate in areas that are used at dusk or dawn. In others, grasslands may be appropriate in tandem with tree planting, such as incorporating native grasses beneath more open tree canopies or in adjacent patches.

While the results of this study are limited to Johannesburg, the approach of contrasting native and novel ecosystems in this paper is relevant to other urban areas. This study expands the Global South literature on NbS, which is critical since much of the literature on urban NbS has been situated in the Global North (Kato-Huerta & Geneletti, 2022; Kowarik, 2011; McPhearson et al., 2025; Rega-Brodsky et al., 2022). Further, while there has been some research on individual components, to our knowledge the contrast between native and novel ecosystems in one urban system has not yet been quantified. Our comparative approach, which explicitly contrasts native and novel ecosystems in a single urban context, could be applied in other cities to inform context-specific NbS design.

5.4.4. Limitations and next steps

A clear limitation of this study is a reliance on correlation to infer a link between ecosystems and climate, biodiversity and well-being. We can only identify associations between variables but cannot establish causal relationships. Our temperature data rely on land surface temperature, which may inflate temperature exposure (Venter et al., 2021). Further, the MODIS dataset is at a 1 km resolution and measures radiative surface temperature, which may be dominated by canopy foliage. Thus, it may not capture micro-scale air variation at a canopy or street level, nor the cooling that may occur beneath the tree canopies (Venter et al., 2021). While this was appropriate for our ward-level analysis, future work may consider finer scale temperature data. We did not explicitly control for other drivers of LST variation, such as elevation, local climate gradients and wind patterns, and impervious cover, which could partially explain some of the variation we attribute to ecosystem type. Indeed, the proportion of sealed to green surfaces in each ward may also explain the difference between native and novel temperatures, but this data was not available.

Potential errors in the SANLC land-cover dataset may create inaccurate results, as misclassifications of parts of the city could lead to biases in the model coefficients. In addition, the use of dominant land cover per ward may over-simplify the results, losing valuable intra-ward heterogeneity. Future research could model land cover as a continuous variable for more nuanced results. The biodiversity intactness index is partly derived from land-use categories and may therefore be correlated with land cover. Alternative biodiversity indicators (like species richness) or field-based surveys could be explored in further studies. For human well-being, we focused on food, fuel and water. Future studies could consider other ecosystem services like flood protection, air quality regulation, or recreational activities. We used the 2011 national census for our ecosystem services and justice metrics, as the 2022 iteration was not yet available at a ward level. Further research could incorporate the more recent census results. That being said, our justice results were limited by the use of ward-level data and dominant race categorisations, which may, like land cover, mask intra-ward heterogeneity. While this scale was appropriate for our analysis, future research may consider more nuanced, fine-scale socio-economic and racial data or indicators. Lastly, this methodology could be applied to other cities with differentiated native and novel ecosystems.

5.5. Conclusion

We find that areas dominated by native and novel ecosystems in Johannesburg are associated with lower mean summer land surface temperatures and higher biodiversity intactness compared to areas dominated by urban-industrial ecosystems, and that native-dominated wards also have higher composite provisioning ecosystem service use. We found that inequities in per capita income mapped onto differences in mean summer land surface temperatures, such that the wealthiest in the city are likely to live in the coolest wards. Although our results rely on correlation and cannot prove causal relationships, the associations largely align with empirical evidence from the broader literature. Our findings extend NbS theory by demonstrating the importance of grounding NbS within its socio-ecological context, particularly in systems with differentiated native and novel ecosystems. They also illustrate how ecological patterns and socio-economic inequities shape NbS benefits in a Global South context, underlining the need for integrated frameworks to assist in policymaking. However, further work is required to identify causal mechanisms that can explain how the outcomes of nature-based solutions might vary between native and novel ecosystems and apply this framework in other urban settings. In a city like Johannesburg with budget constraints, nature-based solutions can be a cost-effective mechanism of pursuing multiple targets at once. Increasing native and novel vegetation in wards with high levels of urban-industrial cover would be beneficial for climate, biodiversity, and human well-being outcomes. In Johannesburg, this implies prioritising the protection and expansion of extant native grasslands, maintaining (and, where appropriate, expanding) the urban canopy, and targeting NbS projects in hotter, lower-income wards, as part of an integrated, ecologically appropriate, and just urban planning strategy.

Author contributions

CH: Conceptualisation, data curation, formal analysis, visualisation, writing – original draft, writing – review and editing; ZV: Conceptualisation, methodology, writing – review and editing; NS: conceptualisation, methodology, writing – review and editing.

6. Conclusion

'Tain't what you do (it's the way that you do it) – Ella Fitzgerald

Mitigation is necessarily organised around carbon as the central metric. At present this is manifest in the net zero goal of balanced carbon accounting. Such a focus, however, may miss a wider context with concomitant systemic interactions, ecological impacts, and distributive consequences (Aguirre-Gutiérrez et al., 2023). This thesis aimed to explore mitigation as part of a larger governance and socio-ecological context, in contrast to current carbon-centric approaches. Failing to take the wider backdrop into account can have negative impacts for multiple facets of the relevant systems, including the mitigation activity itself.

6.1. Main findings

Before drawing out cross-cutting learnings from the dissertation, this section offers a short résumé of the main findings of the three research papers. **Chapter 3** created a novel typology for climate risk management, responding to the current landscape of largely siloed risk management techniques. This typology integrates mitigation (emissions reductions and greenhouse gas removal) within a broader system of interventions that work together to decrease risk (and unavoidable loss and damage) overall.

Chapter 4 was an empirical study of the extent to which global forestation efforts are sited in ecologically appropriate areas. We found that a substantial majority of global forestation projects are not located in forest-dominated ecosystems, with particularly high instances of misplacement on the African continent.

Finally, **chapter 5** provided a local-level empirical study. Urban nature-based solutions (NbS) are a popular climate mitigation tool with biodiversity and human well-being benefits. However, there is limited evidence on how to design NbS where there are contrasting native and novel ecosystems. Considering Johannesburg, South Africa, we found that native (grassy) and novel (treed) ecosystems were associated with cooler and more biodiverse areas of the city, compared to more urban-industrial systems. Higher income was associated with cooler and more biodiverse areas of the city.

6.2. Carbon-centric mitigation as the norm

Chapter 2 discussed that, as mitigation interventions are driven by the global goal of net zero that is embedded in the Paris Agreement, they necessarily focus on carbon as the main metric of success. This enables simplification, quantification, and standardisation of the problem space, which allows for climate change to be made 'governable'. However, because carbon emissions exert a global impact, this standardised metric can result in uncoupling mitigation from its locality, failing to take into account the local social and ecological context.

However, mitigation projects, particularly those relating to the use of nature, must ultimately happen *somewhere* rather than the global *nowhere* (Borie et al., 2021). Therein lies the rub. In the rendering of global knowledge to the local, the mitigation project must enter an existing context, with social, governance, and ecological systems. The dynamics of this context may be insufficiently taken into account where carbon is the sole overriding metric for success. How a purely carbon-centric approach may go wrong was explored in each of the three papers.

6.3. What could go wrong

The focus of mitigation on a single global good, carbon (or, more accurately, greenhouse gas) reduction, can result in a disconnect from the wider system in which it is situated. **Chapter 2** suggested a number of ways in which this could happen, which have been identified in the literature; for example, forestation projects may lead to changes in water availability, and dispossession of land. This can impact the mitigation effort itself: decreased water availability can undermine the persistence of the forestation, while dispossession of land can create negative public sentiment. This dissertation offers three more instances in which a siloed focus on mitigation may have negative ramifications.

Chapter 3 considered how mitigation may be disconnected from a wider system of other climate risk interventions, such as adaptation. This system has interactions between interventions and risk thresholds, which may not be adequately taken into account if only carbon is focused on. Ignoring these interactions and thresholds could mean that friction between different types of interventions may decrease the amount of risk that be lowered overall.

The results of the study in **chapter 4** suggested that a carbon-centric approach to mitigation may lead to disconnection from local ecological systems. Forestation is currently the leading carbon dioxide removal method, used for governmental and corporate mitigation targets (Net Zero Tracker, 2025; S. Smith et al., 2024). There has been some evidence that carbon-centric approaches act as an incentive for tree planting, potentially over more ecologically aligned approaches (Agarwal et al., 2025; Cho et al., 2025; Funk et al., 2014; Rooney & Paul, 2017). Concerns have been raised in the literature that the siting of these projects may ignore the relevant ecological context (Bond et al., 2019; Loft et al., 2024; Parr et al., 2024; Ratnam et al., 2020; Stevens & Bond, 2024; Veldman et al., 2019), given that in some areas abiotic constraints inhibit widespread forest while in others disturbances have caused ecosystems to evolve to be more open (Bond, 2008; Luedeling et al., 2019; Ratnam et al., 2011).

We found that across a database of half a million verified forestation projects, more than half of planting sites were situated in non-forest ecosystems as identified by all three frameworks. Further, a substantial amount of forestation area was found to be situated in open ecosystems, like savannas and woodlands, which are maintained by disturbance regimes (Staver et al., 2011, 2021). Planting trees in these types of ecosystems means that the mitigation effort (forestation) must invariably interact with existing ecosystem structures and associated natural disturbances.

Introducing trees into such a system for the benefit of carbon, and without taking into account the wider ecological system, may result in unintended consequences or indeed the failure of the goals of the projects themselves. The consequence of this is that more forestation projects for carbon purposes may not be universally beneficial.

While not central to the results, **chapter 4** also implies that ecological misplacement may shift burdens and risks onto particular places and thus communities (Redvers et al., 2025). For instance, our results show that a large share of the misalignment occurs in Africa, a continent where other scholars have highlighted the inequitable benefit sharing and 'carbon colonialism' attached to mitigation projects (Fisher et al., 2018; Mulenga, 2025). Forestation in inappropriate systems, for example, may decrease the provision of ecosystem services, which many people rely on for basic needs (Fedele et al., 2021; Prangel et al., 2023; Skhosana et al., 2025).

Chapter 5 shows that carbon-centric governance could lead to a disconnection from the local ecological and social systems. There has been extensive interest in the carbon sequestration offered by Johannesburg's urban trees (a novel ecosystem), with attendant calls to expand the canopy (City of Johannesburg, 2021a; Lembani, 2015; Naidoo et al., 2022; UNFCCC, 2023; Van Staden, 2020; Zakwe, 2025). However, afforestation in the city misses the biodiversity, ecosystem service, and temperature benefits that the native grassland ecosystem could also bring. It may also miss public sentiment towards trees in the city, which may be associated by some residents with higher rates of crime (S. Shackleton et al., 2015). Failing to take into account these details may mean decreased public support, or decreased ecosystem services, both of which may have impacts on the longevity of the trees as well as the potential cost outlays of the municipality.

Further, we found that access to existing NbS benefits in Johannesburg is structured by income: the wealthiest in the city live in the coolest areas. This is reflective of Johannesburg's discriminatory policies that located amenities like trees in areas reserved for white residents (Schäffler & Swilling, 2013). This uneven distribution of benefits has yet to be rectified. In Johannesburg, access to benefits reflects historic and existing power imbalances. Failure to include justice concerns can lead to political or societal resistance towards the relevant mitigation intervention (Klinsky et al., 2017; Trimmel et al., 2024).

This thesis shows that mitigation interventions do not act on empty landscapes; instead, they enter already complex systems. As such, outcomes, including those relating to carbon, depend on interactions, feedback loops, path dependencies, and context, rather than solely the design of the mitigation project or policy. Accordingly, an intervention (like forestation) that would make sense in one system (forest), will not in another (grassland). Where there is misalignment between a mitigation project and its host ecosystem or community, the mitigation may be less effective, resilient, or may even fail. Moving beyond carbon-centric governance thus requires integrated, socio-ecological approaches to mitigation.

6.4. Alternative approaches to mitigation that strike a balance between complexity and ease of use

Mitigation design is robust when it is ecologically aligned with the socio-ecological context in which it is located (P. Smith et al., 2022). How to approach ecologically aligned mitigation is explored in **chapter 5**. The results of the chapter suggest that urban climate-nature interventions also need to work with local-scale ecosystem structures and histories, including both native and novel ecosystems, rather than imported ideals of what urban nature "should" look like (Davis & Robbins, 2018; Roman et al., 2021).

Further, justice and questions of distribution should be central to mitigation design (Hageer, 2025). This is because mitigation is already social and distributive (Welton, 2022). **Chapter 3** suggests risk management itself must be evaluated in terms of which risks are reduced, for whom, and at whose expense. The follow-on article (in **Appendix II**) in particular highlights that the process of using the typology set out in **chapter 3** requires users to turn implicit assumptions into explicit design choices. In so doing, it exposes normative and political commitments, enabling scrutiny thereof.

However, situating mitigation within its socio-ecological context is not meant to create another technical issue to fix (Gillard et al., 2016). While this doctorate has shown that it is critical to situate a mitigation measure within its context, a balance must be struck between this process and not creating undue administrative burden. Increasing the complexity of designing and implementing mitigation projects or policies could increase transaction costs and barriers to entry, thus delaying decarbonisation efforts (Fesenfeld, 2025; Klein et al., 2005; Locatelli et al., 2015; Newkirk et al., 2022). Continued climate change will have deleterious effects on the very same ecosystems or communities, suggesting that a balance must be struck.

One of the hallmarks of the net zero project is how targets are set by individual entities (Welton, 2022). This means that the capacity and capability for engaging with the complexity of the engagements between mitigation and its wider context may vary across institutions. Further, within institutions, there may be decisionmakers with interest in integrated perspectives, but who are stymied by a greater institutional aversion to increasing the complexity of mitigation interventions (Rickards et al., 2014).

However, tools like the typology or the approach taken in the Johannesburg example may be a useful mechanism for introducing socio-ecological complexity without increasing cognitive complexity. Instead, it recognises that mitigation interventions generate multiple outcomes simultaneously: climate regulation; biodiversity effects; provisioning effects; risk redistribution; and social and political consequences. Thus, in **chapter 5** we show that native and novel dominated wards differ from urban-industrial wards across temperature, biodiversity, and provisioning service use, with income also associated with cooler wards. **Chapter 4,** meanwhile, shows how carbon-led mitigation can conflict with biodiversity, ecosystem structure, and resilience when ecological context is ignored. **Chapter 3** formalises the idea that interventions should be analysed through broader risk consequences, not single metrics. Making design choices explicit allows for contestation and debate around plural outcomes. Carbon can thus become one outcome among several, not the sole measure of mitigation success.

6.5. What this thesis contributes

Across the three papers, and this synthesis chapter, this thesis contributes across conceptual, empirical, and practical dimensions. Conceptually, this thesis reframes mitigation as embedded in withing larger governance, social, and ecological contexts, rather than a narrow carbon management exercise. This is then demonstrated across empirical scales: conceptual framework; global forestation governance; and urban NbS. Finally, the thesis suggests that policy should be assessed using criteria such as systemic integration, ecological fit, plural socio-ecological outcomes, and justice.

6.6. Limitations and future research avenues

This thesis has been limited by a number of factors. Some of these are explored in the relevant chapters to which they pertain. However, there are broad philosophical limitations associated with each paper that are not addressed in each.

For example, Stirling (2026) criticises **chapter 3** for failing to challenge existing power dynamics, in particular by presenting the typology in the language of risk (what Stirling calls 'technocratic scientism'). In presenting the typology in this framing ('risk'; 'management'), there is a risk that the tool is not accessible to a broader range of users, despite the intention that it be used by as wide a set of stakeholders as possible. While our follow-up article, in **Appendix II**, clarifies that the use of the typology is not just for quantitative calculations of risk but also for qualitative exercises, this issue taps into a large point about the useability of certain climate framings outside of particular disciplines, as well as what types of knowledge are valued (David, 2024; Jebeile & Roussos, 2023).

This thesis is broadly critical of the global nature of much of the mitigation apparatus (explored most specifically in **chapter 2**). However, in **chapter 4**, local forestation projects are aggregated to a global (and continental) level. This is certainly useful, clearly illustrating the scale and systemic nature of the misplacement of projects. It does, however, reinforce existing knowledge (and thus power) production systems (Turnhout et al., 2016).

Chapter 5 is centred on Johannesburg but all research and writing was carried out in Oxford. Oxford is intimately connected with the British Empire (Symonds, 2011), and the production of knowledge in this city about a former colony without field work could be construed as a perpetuation of historic power imbalances. One of the central tenets of the environmental justice movement is that communities should speak for themselves (Cole & Foster, 2001). While the paper was written by a native of Johannesburg, it did not include the voices of any communities in the city. Further research would rectify this omission.

Similarly, out of the scope of this thesis was direct governance or political analysis, which has the potential to add to the studies. In particular, there is a clear avenue on how the results of **chapters 4** could be translated into policy mechanisms going forward.

In addition, the conceptual structure of **chapter 3** was not carried forward into subsequent chapters. Instead, I focused on empirical studies that contributed more broadly to the dissertation's aims. While this offered additional insights, such as the ecological misplacement of mitigation in non-forest ecosystems set out in **chapter 4**, not bringing through the typology did not test the conceptual framework in practice. Further studies operationalizing the typology in an applied setting would be a useful addition.

Chapter 3 may be limited by the high-level nature of the typology, such that mitigation and adaptation wedges may seem to obscure the differing risk profiles, trade-offs, synergies, and interactions between the constituent interventions within each wedge. However, as is discussed further in **appendix II**, the typology is intentionally designed as a simple framework that users situate within their relevant context. The framework could be applied qualitatively, thus bringing together disparate risk management strategies into a common frame. It could be applied quantitatively, in which case this customization requires explicit engagement with the trade-offs and synergies between individual interventions.

The thesis was also partly limited by methodological design. **Chapters 4 and 5**, for example, were cross-sectional and provided a snapshot of both case studies at a certain point in time. Further studies could include a time series element, such as how future climate change might shift where forestation projects might be ecologically appropriate.

The empirical studies also focus on a particular type of mitigation (which is predicated on the use of nature), which may limit its applicability across all forms of mitigation. However, as discussed above, a wide variety of mitigation techniques have ecological and distributive effects. Future research would consider how these types of mitigation implicate their respective socio-ecological contexts.

Last, it is the author's opinion that any reduction of Johannesburg's remarkable trees, or its distinctive veldt, into a metric is regrettable (Wijsman et al., 2025).

6.7. Final conclusion

A mitigation policy, plan, or project does not only affect how much carbon is reduced or removed. Instead, it can alter ecosystems, or the distribution of benefits and burdens. **Chapter 3** shows that climate action must be understood through interacting portfolios and risk pathways, not single instruments. **Chapter 4** illustrates what happens when mitigation is operationalised too narrowly (forestation projects are ecologically misplaced). **Chapter 5** illustrates a mechanism for considering how a mitigation project may be situated within its social and ecological context.

These research papers show that mitigation cannot be adequately understood through carbon alone, as well as suggesting mechanisms for integrated approaches. Carbon as a metric is critical, given that a surfeit of carbon in the atmosphere is the main driver of climate change. However, focusing too narrowly on it can miss risks and interactions relating to the wider system within which the mitigation effort is situated. Mitigation interventions exist within governance, social, and ecological contexts and are dependent on the processes and properties of host ecosystems and communities. A longer, wider view recognises that these interactions could have consequences for the mitigation intervention over time, and thus are required for durable mitigation. Robust mitigation therefore requires approaches that are integrated, ecologically appropriate, and attentive to justice. After all, as Ella Fitzgerald sang, "'[it] ain't what you do (it's the way that you do it)."

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Appendices

Appendix I

GenAI use declaration

ChatGPT-5.3 Edu, published by OpenAI and available at https://chatgpt.com/, was used in this dissertation to troubleshoot RStudio and Google Earth Engine code. I also used ChatGPT to assist in thinking through the structure of certain sections, by prompting it to ask a series of questions that would make me think it through. Finally, I used it for high-level comments on drafts as if it were my co-author (e.g. 'comment as if you were Nicola Stevens'), so as to pre-emptively improve the draft before sharing. Google Scholar Labs, published by Google and available at https://scholar.google.com/scholar_labs, was used to assist in the literature review process (as a more advanced search tool).

Appendix II

A climate risk management typology: Towards a common framework for co-creation

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In the article, *A climate risk management typology: Integrating approaches to reduce risk*, a stylised climate risk management typology is presented that is deliberately generic but not degenerated. The typology is rendered as simply as possible without omitting key factors, while deliberately ensuring that the framework can be expanded by the user.

That simplicity is pitched at a level that still retains the high-level factors typically salient in climate risk management: emissions reductions, removals, adaptation, loss and damage, and other interventions. We believe that these factors ought to be explicitly included in any complete climate risk management analysis that addresses both the risks of climate change and the risks of responses to it, especially in relation to climate overshoot (Dyke and Bissett 2025, 2). The framework is simple in form, but it does not avoid complexity; since scenarios are represented in terms of risk, they can only be constructed through engagement with impact baselines, responses, outcomes, and the uncertainties, likelihoods, and interactions that arise across physical, socioeconomic, and political dynamics.

This is a core approach of the typology: simple by default, complex as required, which is useful for three connected reasons.

First, simplicity makes the framework usable across a wide range of settings, from the pub to the boardroom to COP. A typology that can be understood and worked with by different actors allows more people to participate substantively in the co-creation of scenarios. That matters because the quality of climate risk analysis depends not only on technical sophistication, but also on whether it is capable of incorporating perspectives beyond disciplinary, technical, and academic silos. In this sense, simplification is not a retreat from rigour, but is a way of widening participation and democratising both the use of the tool.

Second, the typology permits both qualitative and quantitative use; risks do not need to be quantified for the framework to be useful. One of its functions is to bring different actors, concerns, and approaches into the same frame, including areas that are difficult or impossible to quantify. At the same time, the framework can motivate and structure quantitative analysis. A risk-native framing encourages modelling that attempts to resolve uncertainties where possible, while discouraging the omission of major factors simply because they are uncertain. In other words, uncertainty is not treated as a reason to exclude something from analysis.

Third, the typology obliges users to situate the framework in their own context, and to engage with the "inconvenient complex unruliness of climate politics" (Stirling 2026, 3). One purpose of the typology was precisely to address the real world eventualities and possibilities that may undermine pathways that are often depicted as immutable lines extending solidly for decades and centuries. It would be infeasible for any broad framework to provide deep treatment of all the debates and uncertainties associated with every factor across all scenarios, scales, and regions. Nor would it be appropriate for the structure of the tool itself to predetermine what must be salient in every case. Instead, the typology provides a scaffold within which users must work through the full gamut of complexities relevant to them. Its value lies in requiring a more holistic consideration of the range of factors and eventualities relevant to a given case, including those often omitted when overshoot discourse is organised in silos.

This point is best understood through the way the framework turns implicit assumptions into explicit design choices. The scope of the reason for concern, for example, can vary hugely, including whether derailment risks are included within the risks being considered (Dyke and Bissett 2025), and even whether the relevant risk is understood as applying only to humans or also to other species (Tschakert et al. 2021). Likewise, if a user chooses to employ climate tipping points as risk thresholds, that decision immediately requires engagement with qualitatively different risk landscapes involving irreversibility and abrupt changes, and with contestation around the thresholds themselves and their significance (Ritchie et al. 2021; Kopp et al. 2025).

The same is true of mitigation choices: afforestation, reforestation, and forest management are currently the most widely deployed carbon dioxide removal approaches (Smith et al. 2024), but tree planting for carbon remains contentious because of biodiversity impacts, changes to albedo, and the impermanence of sequestration (Bond et al. 2019; Fleischman et al. 2020; Kirschbaum et al. 2024). Adaptation, similarly, cannot simply be assumed to reduce risk without further examination. Its role may depend on debates about the limits of adaptation, the potential for learning over time, and the extent to which societies can align normatively on what they are trying to preserve (Driessen and van Rijswick 2011; Callahan 2025). In each case, questions that might otherwise remain buried within climate analysis become visible as choices that must be made.

This is also why a systematic typology does not necessarily operate as a form of technocratic scientism. Stirling's critique is important precisely because it draws attention to the danger that political commitments may be hidden behind the neutral authority of science. But the typology is designed to do the opposite. By forcing users to specify the risk function, define thresholds, choose scenario boundaries, and decide which harms, uncertainties, and interactions are important, it strips away the illusion of neutral, objective modelling. It makes clear that even highly systematic analysis rests on prior judgements about salience, tolerability, responsibility, and value. In that sense, the framework does not depoliticise climate risk management. It exposes the normative and political commitments already embedded within it, and makes them available for scrutiny, contestation, and revision.

This is why the typology does not prescribe which risks matter most, which political or physical complexities deserve priority, what the objective of a scenario should be, what decision calculus should be used, what degree of risk aversion is appropriate, or what level of climate risk different stakeholders should accept. Rather, it gathers into one frame the high-level factors already present in climate discourse and encourages broader exploration of the combinatorial scenario space. Its normative commitment is not to a specific outcome, but to the value of more holistic and integrated consideration under present planetary conditions.

It follows that the original article does not itself evaluate, analyse, or assess climate risks in any substantive sense; the example portfolios are illustrative and non-evaluative. They show how the high-level factors in the typology can be used to structure scenario variations already present in high-level international climate discourse. In that sense, they broadly reflect the current state of discussion rather than settling it. Their purpose is to offer a common language for discussing qualitatively distinct, but not quantitatively assessed, responses to overshoot.

This common language matters because labels such as Plan A and Plan B are already in circulation (Corry 2017; Pierrehumbert 2019; Davenport 2021; Browne 2022; Fuglesang and Hassler 2023). That reality itself creates a need for greater terminological coherence, so as to reduce risks of misunderstanding and miscommunication. Common language can provide clearer focal points around which research, policy, and public communities can convene. There is precedent for this in climate politics (Randalls 2010; Jaeger and Jaeger 2011; Cointe and Guillemot 2023); global temperature targets emerged as a simplified, co-created, commonality from the wider complexity of the climate problem. That simplification helped democratise discourse, even as it also risked myopic orientation towards an intermediate proxy goal. Our typology aims to retain the benefits of simplification without collapsing into excessive reductionism.

A second major issue raised in the responses concerns uncertainty. Here again, the typology is intended not to dissolve uncertainty but to require engagement with it. Because the framework is cast in terms of risk, some structuring of likelihoods and uncertainties is essential. This occurs through the choice of risk functions, the placement of thresholds, and the construction of scenarios. These choices may be informed by quantitative estimates, qualitative judgements, or both. Not all uncertainties are alike, and not all are equally tractable. Distinguishing epistemic, structural, aleatoric, ontological, and normative uncertainties across the wedges, and both physical and social domains, can help illuminate where uncertainty-derived risk is most significant. Rather than treating uncertainty as a residual complication, the framework makes it part of the architecture of analysis.

This is particularly important in relation to questions around solar radiation management, mitigation deterrence, and termination shock. If a scenario includes SRM, the user must represent not only any possible risk-reducing effects, but also the conditions on which those effects depend and likelihoods of their continuation: sustained implementation, governance stability, compliance, and the balance of social and political aversion and tolerance. Likewise, if mitigation deterrence or broader derailment risks are judged salient, these can be built directly into the scenario structure or the risk function itself (Dyke and Bissett 2025). In this way, the typology does not treat such concerns as external objections appended after the main analysis. It requires them to be incorporated into the same risk landscape from the outset.

More broadly, the framework requires engagement with the messy realities underlying the mitigation, adaptation, removal, or intervention wedges commonly used in high-level climate analysis. Their apparent simplicity often obscures the plurality of assumptions, trade-offs, uncertainties, and value judgements they contain. The typology does not pretend to eliminate that messiness. Rather, by reducing the problem to essential parts at the outset, it creates space for users to rebuild a fuller scenario in a way that could make those tensions more tangible.

The value of a management framing depends on how it is used. A management orientation is useful only if it remains attentive to blind spots, interactions between interventions, and the possibility that actions do not proceed as intended because of non-compliance, political constraints, social pressures, or unintended side effects. The typology encourages exactly this by requiring consideration of how interventions combine within one framework, and what their joint implications may be. By contrast, approaches organised around a single physical variable may narrow the scope of analysis and make it harder to see interactions, trade-offs, and wider system effects. The point of the typology is to broaden that scope and support a more integrated view of climate risk management.

For this reason, risk is also an appropriate integrating metric. In some contexts, it is more useful than monetary value or temperature. Many climate risks are not financially representable. Temperature, although useful, functions in some respects as a proxy and instrumental sub-goal for risk itself. Risk, by contrast, directly incorporates a wider multiplicity of dynamics and inherently requires engagement with uncertainty (Webster et al. 2012). It also has communicative value: risk is something people experience more readily than large scale indicators such as global mean temperature. The tool is therefore intended to structure thinking, ensure comprehensiveness, and connect different actors through a shared risk-based language.

This orientation towards risk is not an invention of our framework. It reflects the foundations of international climate governance. The Paris Agreement sits within the United Nations Framework Convention on Climate Change, whose objective is the "stabilization of greenhouse gas concentrations […] at a level that would prevent dangerous anthropogenic interference with the climate system" (United Nations Framework Convention on Climate Change 1992, art. 2). That objective is coupled with principles of equity and common but differentiated responsibilities (art. 3.1), and with a precautionary approach (art. 3.3), according to which lack of full scientific certainty should not justify delaying measures where there are threats of serious or irreversible damage. The deeper challenge of climate governance is therefore not simply to achieve a particular temperature outcome. It is to identify and avoid what counts as dangerous climate change under conditions of uncertainty and plural values (Schneider 2001; Dessai et al. 2004; Oppenheimer 2005).

The typology is intended to operate within precisely this space. It does not prescribe what should count as dangerous, what level of risk is acceptable, or which responses are preferable. Rather, it provides a structure within which such questions can be made explicit and considered in relation to one another. By recasting scenarios in terms of risk rather than relying solely on global mean temperature as a proxy (Hulme 2018), the framework foregrounds the locally contingent, value-laden, and uncertain character of climate impacts, while also allowing more integrated consideration of trade-offs and interactions between responses.

Finally, this returns us to the role of simplification. In many climate analyses, engagement begins with high complexity, with different actors adopting different framings, variables, and assumptions, which are not mutually comparable or representable. The result is often that analyses proceed in parallel rather than in dialogue. The typology takes the opposite approach. By beginning from a deliberately simple shared structure, it establishes a common conceptual root from which more detailed analysis can develop. Complexity is not removed; it is introduced as required through the specification of risk functions, thresholds, and scenarios. Starting from a common root helps different perspectives remain commensurable as complexity increases. It is this combination – simple by default, complex as required – that underpins the utility of the typology as a tool for navigating the interconnected, uncertain, and value-laden nature of climate risk management.

Bibliography to Appendix II

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Appendix III

Supplementary materials to Chapter 4

***Table S 1**: Large-scale forestation projects excluded during data cleaning and the reasons for exclusion.*

Site ID	Derived planting site size (km^2)	Project description (reported)	Reason for removal
464655	3,271,096.158	Grouped Project for Reforestation on Degraded Land and Fallow Agricultural Lands (hereafter referred as the project) s located in geographical boundary of India. The project is aimed at establishing forests on degraded lands and unused non-forest lands at various locations of India. The grouped project is being implemented by EKI Energy Services Ltd.	Location too vague.
980922	3,264,013.935	The proposed grouped project is an afforestation, reforestation, and revegetation (ARR) initiative aimed at increasing vegetation cover by planting native tree species on agricultural fallow lands and/or degraded farmlands. The proposed project activity is a voluntary project activity being implemented by LTIMindtree. The project is grouped project activity with the geographic boundary of India. The first instance of the grouped project activity is implemented in 2128 ha distributed across 9 states, Uttarakhand, Uttar Pradesh, Madhya Pradesh, Maharashtra, West Bengal, Odisha, Telangana, Andhra Pradesh and Karnataka) of India. The second instance is expected to cover around 2000 ha in the above-mentioned states or in any other states of India.	Location too vague.
1283266	1,106,446.568	We are seeking carbon credit offtake partners for our pilot project. Project highlights: - Location: Atlantic Forest biome, South eastern Brazil. - Size: ~3,000 Ha. - Volume: ~32,000 TC02e average over 12 years. - Offer: Carbon credit offtake agreement. Payment on delivery. 12 year term (different periods can be negotiated). - Price: Reserve price of 60USD/TC02e / verified carbon credit. Fixed price throughout offtake agreement.	Location too vague.
444530	946,651.820	The project "Mahogany Planation in India" is a grouped afforestation activity in form of agrisilvicultural system that intends to satisfy multiple Sustainable Development Goals through project activity ("Project or Project Activity"). The project is implemented on multiple scattered small land parcels within the 65 districts (defined by Government of India, as on 28 June 2019, Project Start date, across the States of Maharashtra, Madhya Pradesh, Karnataka, Gujarat and Telangana, in India.In pre-project scenario, the agricultural lands owned by farmers are used for traditional farming and shall be used for Project Activity with continuation of traditional farming. The intent of PP is to create a stable sustainable income source for the farmer who is solely dependent on traditional subsistence driven farming. The Project Activity is implemented through multiple consultation and engagements between PP and the relevant stakeholders including but not limited to farmer, local governance bodies and administrations. The stakeholders involved in	Location too vague.

Site ID	Derived planting site size (km^2)	Project description (reported)	Reason for removal
		the Project Activity are mutually bound by legal framework in compliance with all the local and national laws and regulations. The proposed Project covers multiple scattered small landholdings privately owned by various farmers aggregating to 15,000 hectares' land across the Project Area. PP intends to complete the plantation activity in 15,000 hectares by the year 2024, which shall create direct positive impact on livelihoods of more than 25,000 farmer families, i.e. 100,000 individuals and indirect positive impacts on large number of individuals involved in Project Activity.	
1289019	770,181.744	N/A	Not enough information.
996428	498,707.882	India's agricultural landscape, shaped by smallholder farming and diverse crop cultivation using traditional methods, contributes approximately 14% of the country's gross greenhouse gas (GHG) emissions. Afforestation emerges as a pivotal strategy to address environmental challenges and foster resilient ecosystems amid issues such as deforestation, habitat loss, and climate change. It plays a crucial role in restoring degraded land, enhancing biodiversity, and mitigating the impacts of climate change. Cropcity Agro Carbons Private Limited (CAPL) leads an Afforestation, Reforestation, and Revegetation (ARR) project in Karnataka and Maharashtra states, partnering with KMS Private Limited for carbon credit advisory services. The initiative mobilizes farmers to cultivate fruit-bearing trees like Indian blackberry and Jackfruit on fallow agricultural land. Afforestation projects mitigate climate change by sequestering carbon dioxide through tree planting, contributing to greenhouse gas emission reductions.	Location too vague.
19736	307,851.272	The afforestation project is a vital initiative that combats climate change by promoting the plantation of bamboo with landowners and communities. By absorbing carbon at an exceptional rate and preventing soil erosion, bamboo provides a sustainable and long-term solution for mitigating the effects of climate change. The project not only helps to create a healthier environment but also provides a valuable source of income for local communities, making it a truly impactful initiative. Furthermore, bamboo's versatility as a food source for wildlife, a sustainable material for construction and furniture, and a potential source of traditional medicine, makes it an invaluable resource for communities in tropical regions.	Location too vague.
1282921	233,654.492	## Background Forest restoration and rehabilitation are effective tools to reconnect isolated forest patches. With support from HP, WWF is working with local partners to promote restoration of priority areas in the Upper Parana and Serra do Mar coastal ecoregions within the Atlantic Forest of Brazil, with a focus on biodiversity corridors and water provisioning. ## Scope Of Work With support from HP, WWF will restore more than 1,300 acres (550	Location too vague.

Site ID	Derived planting site size (km^2)	Project description (reported)	Reason for removal
		hectares) of land in Brazil's Atlantic Forest region to native forest. To accomplish this goal, WWF will collaborate with local and regional partners to build a strong foundation so that communities are engaged and benefiting from the restoration and that these reforested areas thrive over the long term. HP's financial investment in this region will also complement forest restoration that Sylvamo, HP's supply chain partner, already supports in [the Mogi Guaçu River basin of the Atlantic Forest](https://explorer.land/p/project/mogi-guacu-river-basin).	
855145	214,414.857	Proventus Grouped Project has been developed under the Verified Carbon Standard (VCS) following CDM AR-ACM0003 Methodology and aims to promote investments among small - and medium- sized landowners by new commercial forest plantations. The project is based on changing the use of low productivity land to sustainable forest production systems, which will increase the forest cover and promote remnant natural forest improvement generating a landscape of biological corridors that bring about financial, social and environmental services.	Not enough information.
944910	202,881.152	A forest carbon credit project to create greenhouse gas (GHG) emission reductions and removals by converting privately owned operational forest lands to protected forest lands. The project will be implemented following the Verified Carbon Standard (VCS) VM0012 – Improved Forest Management in Temperate and Boreal Forests (LtPF), v1.2 methodology. The project is located in coastal British Columbia, Canada. This forest carbon project area is non-contiguous, with parcels located throughout Vancouver Island, Cortes Island, and Haida Gwaii. Old forests, ecologically significant areas, and culturally important areas were targeted for the forest carbon project.	Location too vague.
1268650	165,205.908	The Northern Region of Zambia is rich in natural resources, with large lakes, rivers, and fertile land for agriculture. It has vast forests, covering about 40-50% of the area, but faces growing threats from deforestation. The main causes are charcoal production, slash-and-burn farming, and illegal logging. This deforestation leads to loss of biodiversity and impacts climate resilience. Key efforts to address these challenges include: climate-smart agriculture, reforestation, and renewable energy projects. The region is home to 30-40% of Zambia's biodiversity, with protected areas like Nsumbu National Park safeguarding species like the Shoebill Stork. Despite challenges, local initiatives are working to restore and protect these vital ecosystems.	Location too vague.
996379	162,976.913	The proposed agroforestry project is an afforestation, reforestation, and revegetation (ARR) initiative aimed to promote sustainable agriculture by planting native agroforestry species in Andhra Pradesh and Karnataka states in India and it is being implemented by GKF Agroforestry Pvt. Ltd. The project includes planting of	Mismatch between reported size (11,000 hectares) and derived size

Site ID	Derived planting site size (km^2)	Project description (reported)	Reason for removal
		native and locally adapted agroforestry species in 11,000 hectares of land	(~16.5 million hectares).
1289023	157,020.804	N/A	Not enough information.
983446	129,527.046	The Project seeks to develop an Afforestation, Reforestation and Revegetation (ARR) VCS grouped project across Latvia and Lithuania.	Location too vague.
980653	112,126.257	This is an agroforestry project on minor agriculture crop land and barren land aims to promote sustainable agroforestry practices across Telangana state of India. By planting diverse range of trees the project seeks to empower smallholders and tribal communities, increase tree cover, enhance income, conserve biodiversity, improve soil health, and generate emission reductions. The targeted farmers are small landholders and the land earmarked for plantation activities is privately owned. Encompassing a total plantation of 6,000,000 trees in 10,000 ha of land, the farmers have undertaken tree plantations, primarily featuring Mangifera indica (mango), Santalum album (sandalwood), Psidium guajava (guava), Tectona grandis (teak), Azadirachta indica (neem), Swietenia macrophylla (mahogany), Casuarina equisetifolia (horsetail she-oak), Dalbergia sissoo (Indian rosewood), Aquilaria malaccensis (agarwood), Phyllanthus emblica (amla), Punica granatum (pomegranate), and Manilkara zapota (sapota) during the years 2021, 2022, and 2023. The districts included in the project area are : Vikarabad, Ranga Reddy, Mahabub Nager, Narayanapeta, Sanga Reddy, Jogulambha Gadwal, Wanaparthy and Medchiel.	Mismatch between reported size (10,000 hectares) and derived size (~11 million hectares).
1266699	95,990.997	The Pastoralist Economic and Social Advancement (PESA) is a non-governmental, non-political, pro-development organization dedicated to improving the quality of life for pastoralist communities in Tanzania. This proposal outlines our significant contributions to nature restoration and grassroots efforts, aligning with the objectives of the RestorLife Award.	Not enough information.
18345	95,743.071	The Malawi Bamboo Afforestation Project will distribute seedlings of giant bamboo, Dendrocalamus asper, to individual agricultural homesteads in rural Malawi. The project will also provide educational support and outreach for the long-term care of bamboo clumps. The project is located within the political boundary of the Republic of Malawi. The project will generate GHG emission reductions by sequestering carbon in live above- and belowground biomass. When individual bamboo clumps reach the end of their lifespan, they will be replaced by a new clump, ensuring carbon remains sequestered over the project lifetime. Smallholder agricultural homesteads are widespread across Malawi. A variety of trees are grown on farms, in gardens, and around homesteads to serve as fuel, timber, and other uses. However, fast-growing giant bamboo is not grown in Malawi at a large scale. Estimated	Location too vague.

Site ID	Derived planting site size (km^2)	Project description (reported)	Reason for removal
		annual average GHG emissions reductions: 174,301 tCO2e Total GHG emission reductions: 5,054,739 tCO2e	
348503	94,039.210	The Mindanao Tree Planting Program for our Climate and Communities (MinTrees) is a de-centralized, long-term rural sustainable development project focused on smallholder agriculture and forest management. It reaches a large number of beneficiary households and municipalities in the Mindanao region of the Philippines. It involves activities of Afforestation, Reforestation and Revegetation (ARR), especially agroforestry systems and ecological reforestation. MinTrees is scaling up according to a long-term roll-out strategy with several instances of project activities in specific areas. Therefore, it is a Grouped Project that defines large parts of the Mindanao region as geographically defined area for the grouped project instances. For example, annual planting waves keep expanding the project area of ARR instances with agroforestry and reforestation activities. Each project instance has and will have its unique specified geographic project area and where applicable reference regions, leakage management areas etc. The project's community objectives are the inclusion and sustainable livelihood development for tens of thousands of rural community households from both indigenous and non-indigenous origins. The project is applying a unique approach to blending or combining several finance streams for rural transformation including sustainable agri-business, soft loan agri-lending, development grants and climate finance. For productive lands of the ARR component, rural communities are assisted with the capacity building and empowerment of their farming groups, cooperatives and clusters. They receive access to a complete and successfully tested package of diversified agroforestry production, training, soft loan for working capital, quality control, extension services and most importantly a clear and specific market access.	Location too vague.
1268436	86,916.677	N/A	Not enough information.
721978	84,658.548	Our project, based in Uganda, focuses on carbon removal by collaborating with private landowners to significantly enhance the country's vegetation cover. Our goal extends beyond carbon removal; we strive to uplift local communities and enrich biodiversity. Through our initiatives, we aim to create a positive impact that transcends environmental benefits, fostering sustainable development and harmony between people and nature.	Location too vague.
4495	73,587.183	500 smallholder cocoa farmers located near the towns of Bafoussam, Douala and Yaoundé (Western Cameroon) will grow higher quality cocoa while improving their land (125 acres) through the intercropping of 500,000 multipurpose trees and fruit trees per year	Mismatch between reported size (125 acres) and derived size (~18 million acres).

Site ID	Derived planting site size (km^2)	Project description (reported)	Reason for removal
980236	58,623.058	The TERAKA program is a comprehensive reforestation and sustainable development initiative in Madagascar, led by subsistence farmers. Inspired by the successful TIST model in East Africa, farmers plant and own trees on their land, receiving training from TERAKA and 70% of carbon benefits. Small Groups of farmers are empowered to combat deforestation, poverty, and drought. By integrating sustainable practices with carbon sequestration, the program supports farmers in reforestation and biodiversity preservation, creating a 'virtual' cash crop from carbon credits. Operating in central and eastern Madagascar, TERAKA aims to plant trees across 6,000 hectares within the initial eight years. This initial phase is expected to remove approximately 1,971,413 tonnes of greenhouse gases over 40 years, while restoring degraded land, improving the livelihoods and well-being of local communities, fostering sustainable employment, and promoting biodiversity.	Mismatch between reported size (6000 hectares) and derived size (~5.8 million acres).
392247	54,683.596	The Grouped Project Sul da Bahia falls under the Afforestation, Reforestation and Revegetation (ARR) category and targeting the following Verra certifications: Verified Carbon Standard (VCS), and Climate, Community & Biodiversity Standard (CCB). The project aims to restore forest formations that are diminished due to the expansion of other land use practices such as overgrazing on extensive cattle ranches, intensive agriculture and silviculture systems. The grouped project is located in the Atlantic Forest biome in the state of Bahia (Brazil). Restoration is attained through forestation activities using native and non-native species. Project activities also include harvesting and establishing agroforestry systems to diversify and maximize project benefits for all parties and communities involved. The overarching goal of the project is to create a biodiverse forest cover that combines carbon capture, biodiversity and community impacts with a diversity of economic functions such as sustainable production of crops and forest products.	Location too vague.
721979	54,544.840	Our project, based in Uganda, focuses on carbon removal by collaborating with private landowners to significantly enhance the country's vegetation cover. Our goal extends beyond carbon removal; we strive to uplift local communities and enrich biodiversity. Through our initiatives, we aim to create a positive impact that transcends environmental benefits, fostering sustainable development and harmony between people and nature.	Location too vague.
1271098	50,971.350	A priority site for GEF 8 in Madagascar	Not enough information.

Table S 2: Number of forestation planting sites excluded during data cleaning, grouped by reason for exclusion.

Reason for exclusion	Number of projects excluded	Percentage of Original Dataset
Blue-carbon or marine related terms	118,580	9.20
Non-forest terms	162,358	12.6
Both blue carbon and non-forest terms	35	0.00272
Projects located in Antarctica or NA	22,543	1.75
Projects matched exact admin area	1,143	0.09
Projects > 50,000km^2	65	0.005

Table S 3: Sensitivity of alignment estimates to polygon overlap threshold. *Values show the percentage of planting sites classified as aligned under different overlap thresholds.*

Threshold	Biophysical forest potential (%)	Predicted natural tree cover (%)	Ecoregion-based forest classification (%)
0.1	10.91	47.73	61.84
0.2	10.80	47.71	61.81
0.3	10.72	47.69	61.78
0.4	10.64	47.68	61.76
0.5	10.58	47.67	61.73
0.6	10.50	47.65	61.71
0.7	10.43	47.64	61.68
0.8	10.35	47.62	61.65
0.9	10.26	47.60	61.61

Table S 4: Categorisation of RESOLVE ecoregions as forested or non-forested (open; mosaic; and treeless). *We include the name of the ecoregion, its relevant biome, its RESOLVE ID number, as well as the category. We categorised them as either forested or non-forested. Non-forested ecoregions were either categorised as open, mosaic or treeless. Definitions of each is set out in methods.*

Ecoregion	Biome	ID	Category
Alaska Peninsula montane taiga	Boreal Forests/Taiga	369	Forest
Central Canadian Shield forests	Boreal Forests/Taiga	370	Forest
Cook Inlet taiga	Boreal Forests/Taiga	371	Forest
Copper Plateau taiga	Boreal Forests/Taiga	372	Forest
East Siberian taiga	Boreal Forests/Taiga	710	Forest
Eastern Canadian forests	Boreal Forests/Taiga	373	Forest
Eastern Canadian Shield taiga	Boreal Forests/Taiga	374	Forest
Iceland boreal birch forests and alpine tundra	Boreal Forests/Taiga	711	Forest
Interior Alaska-Yukon lowland taiga	Boreal Forests/Taiga	375	Forest
Kamchatka-Kurile meadows and sparse forests	Boreal Forests/Taiga	713	Forest
Kamchatka taiga	Boreal Forests/Taiga	712	Forest
Mid-Canada Boreal Plains forests	Boreal Forests/Taiga	376	Forest
Midwest Canadian Shield forests	Boreal Forests/Taiga	377	Forest
Muskwa-Slave Lake taiga	Boreal Forests/Taiga	378	Forest
Northeast Siberian taiga	Boreal Forests/Taiga	714	Forest

Ecoregion	Biome	ID	Category
Northern Canadian Shield taiga	Boreal Forests/Taiga	379	Forest
Okhotsk-Manchurian taiga	Boreal Forests/Taiga	715	Forest
Sakhalin Island taiga	Boreal Forests/Taiga	716	Forest
Scandinavian and Russian taiga	Boreal Forests/Taiga	717	Forest
Southern Hudson Bay taiga	Boreal Forests/Taiga	382	Forest
Trans-Baikal conifer forests	Boreal Forests/Taiga	718	Forest
Urals montane forest and taiga	Boreal Forests/Taiga	719	Forest
Watson Highlands taiga	Boreal Forests/Taiga	383	Forest
West Siberian taiga	Boreal Forests/Taiga	720	Forest
Northern Cordillera forests	Boreal Forests/Taiga	380	Forest
Northwest Territories taiga	Boreal Forests/Taiga	381	Forest
Afghan Mountains semi-desert	Deserts & Xeric Shrublands	807	Treeless
Alashan Plateau semi-desert	Deserts & Xeric Shrublands	808	Treeless
Aldabra Island xeric scrub	Deserts & Xeric Shrublands	91	Treeless
Arabian sand desert	Deserts & Xeric Shrublands	810	Treeless
Araya and Paria xeric scrub	Deserts & Xeric Shrublands	597	Treeless
Atacama desert	Deserts & Xeric Shrublands	598	Treeless
Saharan Atlantic coastal desert	Deserts & Xeric Shrublands	839	Treeless
Azerbaijan shrub desert and steppe	Deserts & Xeric Shrublands	812	Treeless
Badghyz and Karabil semi-desert	Deserts & Xeric Shrublands	813	Treeless
Baja California desert	Deserts & Xeric Shrublands	426	Treeless
Baluchistan xeric woodlands	Deserts & Xeric Shrublands	814	Open
Caribbean shrublands	Deserts & Xeric Shrublands	599	Treeless
Carnarvon xeric shrublands	Deserts & Xeric Shrublands	207	Treeless
Caspian lowland desert	Deserts & Xeric Shrublands	815	Treeless
Central Afghan Mountains xeric woodlands	Deserts & Xeric Shrublands	816	Open
Central Asian northern desert	Deserts & Xeric Shrublands	817	Treeless
Central Asian riparian woodlands	Deserts & Xeric Shrublands	818	Mosaic
Central Asian southern desert	Deserts & Xeric Shrublands	819	Treeless
Central Mexican matorral	Deserts & Xeric Shrublands	427	Treeless
Central Persian desert basins	Deserts & Xeric Shrublands	820	Treeless
Central Ranges xeric scrub	Deserts & Xeric Shrublands	208	Treeless
Chihuahuan desert	Deserts & Xeric Shrublands	428	Treeless
Colorado Plateau shrublands	Deserts & Xeric Shrublands	429	Treeless
Cuban cactus scrub	Deserts & Xeric Shrublands	600	Treeless
Deccan thorn scrub forests	Deserts & Xeric Shrublands	315	Open
Djibouti xeric shrublands	Deserts & Xeric Shrublands	92	Treeless
East Arabian fog shrublands and sand desert	Deserts & Xeric Shrublands	821	Treeless
East Sahara Desert	Deserts & Xeric Shrublands	822	Treeless
East Saharan montane xeric woodlands	Deserts & Xeric Shrublands	823	Open
Eastern Gobi desert steppe	Deserts & Xeric Shrublands	824	Treeless
Eritrean coastal desert	Deserts & Xeric Shrublands	93	Treeless
Galápagos Islands xeric scrub	Deserts & Xeric Shrublands	601	Treeless
Gariep Karoo	Deserts & Xeric Shrublands	94	Treeless
Gibson desert	Deserts & Xeric Shrublands	209	Treeless

Ecoregion	Biome	ID	Category
Godavari-Krishna mangroves	Deserts & Xeric Shrublands	316	Forest
Gobi Lakes Valley desert steppe	Deserts & Xeric Shrublands	825	Treeless
Great Basin shrub steppe	Deserts & Xeric Shrublands	430	Treeless
Great Lakes Basin desert steppe	Deserts & Xeric Shrublands	826	Treeless
Great Sandy-Tanami desert	Deserts & Xeric Shrublands	210	Treeless
Great Victoria desert	Deserts & Xeric Shrublands	211	Treeless
Guajira-Barranquilla xeric scrub	Deserts & Xeric Shrublands	602	Treeless
Gulf of California xeric scrub	Deserts & Xeric Shrublands	431	Treeless
Hobyo grasslands and shrublands	Deserts & Xeric Shrublands	95	Treeless
Indus Valley desert	Deserts & Xeric Shrublands	317	Treeless
Junggar Basin semi-desert	Deserts & Xeric Shrublands	827	Treeless
Kalahari xeric savanna	Deserts & Xeric Shrublands	97	Open
Kaokoveld desert	Deserts & Xeric Shrublands	98	Treeless
Kazakh semi-desert	Deserts & Xeric Shrublands	828	Treeless
Kopet Dag semi-desert	Deserts & Xeric Shrublands	829	Treeless
La Costa xeric shrublands	Deserts & Xeric Shrublands	603	Treeless
Madagascar spiny thickets	Deserts & Xeric Shrublands	99	Treeless
Madagascar succulent woodlands	Deserts & Xeric Shrublands	100	Open
Malpelo Island xeric scrub	Deserts & Xeric Shrublands	604	Treeless
Meseta Central matorral	Deserts & Xeric Shrublands	432	Treeless
Mesopotamian shrub desert	Deserts & Xeric Shrublands	830	Treeless
Mojave desert	Deserts & Xeric Shrublands	433	Treeless
Motagua Valley thornscrub	Deserts & Xeric Shrublands	605	Treeless
Nama Karoo shrublands	Deserts & Xeric Shrublands	101	Treeless
Namaqualand-Richtersveld steppe	Deserts & Xeric Shrublands	102	Treeless
Namib Desert	Deserts & Xeric Shrublands	103	Treeless
Namibian savanna woodlands	Deserts & Xeric Shrublands	104	Open
North Arabian desert	Deserts & Xeric Shrublands	831	Treeless
North Arabian highland shrublands	Deserts & Xeric Shrublands	832	Treeless
North Saharan Xeric Steppe and Woodland	Deserts & Xeric Shrublands	833	Open
Somali montane xeric woodlands	Deserts & Xeric Shrublands	106	Open
Aravalli west thorn scrub forests	Deserts & Xeric Shrublands	314	Open
Nullarbor Plains xeric shrublands	Deserts & Xeric Shrublands	212	Treeless
Paraguaná xeric scrub	Deserts & Xeric Shrublands	606	Treeless
Paropamisus xeric woodlands	Deserts & Xeric Shrublands	834	Open
Pilbara shrublands	Deserts & Xeric Shrublands	213	Treeless
Qaidam Basin semi-desert	Deserts & Xeric Shrublands	835	Treeless
Red Sea-Arabian Desert shrublands	Deserts & Xeric Shrublands	837	Treeless
Red Sea coastal desert	Deserts & Xeric Shrublands	836	Treeless
Registan-North Pakistan sandy desert	Deserts & Xeric Shrublands	838	Treeless
San Lucan xeric scrub	Deserts & Xeric Shrublands	607	Treeless
Sechura desert	Deserts & Xeric Shrublands	608	Treeless
Simpson desert	Deserts & Xeric Shrublands	214	Treeless
Snake-Columbia shrub steppe	Deserts & Xeric Shrublands	434	Treeless
Socotra Island xeric shrublands	Deserts & Xeric Shrublands	105	Treeless

Ecoregion	Biome	ID	Category
Sonoran desert	Deserts & Xeric Shrublands	435	Treeless
South Iran Nubo-Sindian desert and semi-desert	Deserts & Xeric Shrublands	841	Treeless
South Sahara desert	Deserts & Xeric Shrublands	842	Treeless
Southwest Arabian Escarpment shrublands and woodlands	Deserts & Xeric Shrublands	108	Mosaic
Southwest Arabian highland xeric scrub	Deserts & Xeric Shrublands	109	Treeless
St. Peter and St. Paul Rocks	Deserts & Xeric Shrublands	609	Treeless
Succulent Karoo xeric shrublands	Deserts & Xeric Shrublands	110	Treeless
Taklimakan desert	Deserts & Xeric Shrublands	843	Treeless
Tamaulipan matorral	Deserts & Xeric Shrublands	436	Treeless
Tamaulipan mezquital	Deserts & Xeric Shrublands	437	Treeless
Tehuacán Valley matorral	Deserts & Xeric Shrublands	610	Treeless
Thar desert	Deserts & Xeric Shrublands	318	Treeless
Tibesti-Jebel Uweinat montane xeric woodlands	Deserts & Xeric Shrublands	844	Open
Tirari-Sturt stony desert	Deserts & Xeric Shrublands	215	Treeless
West Sahara desert	Deserts & Xeric Shrublands	845	Treeless
West Saharan montane xeric woodlands	Deserts & Xeric Shrublands	846	Open
Western Australian Mulga shrublands	Deserts & Xeric Shrublands	216	Treeless
Wyoming Basin shrub steppe	Deserts & Xeric Shrublands	438	Treeless
Southwest Arabian coastal xeric shrublands	Deserts & Xeric Shrublands	107	Treeless
Arabian-Persian Gulf coastal plain desert	Deserts & Xeric Shrublands	811	Treeless
South Arabian plains and plateau desert	Deserts & Xeric Shrublands	840	Treeless
Arabian desert	Deserts & Xeric Shrublands	809	Treeless
Ile Europa and Bassas da India xeric scrub	Deserts & Xeric Shrublands	96	Treeless
Amur meadow steppe	Flooded Grasslands & Savannas	741	Treeless
Bohai Sea saline meadow	Flooded Grasslands & Savannas	742	Treeless
Cuban wetlands	Flooded Grasslands & Savannas	579	Mosaic
East African halophytics	Flooded Grasslands & Savannas	69	Treeless
Enriquillo wetlands	Flooded Grasslands & Savannas	580	Treeless
Etosha Pan halophytics	Flooded Grasslands & Savannas	70	Treeless
Everglades flooded grasslands	Flooded Grasslands & Savannas	581	Treeless
Guayaquil flooded grasslands	Flooded Grasslands & Savannas	582	Treeless
Inner Niger Delta flooded savanna	Flooded Grasslands & Savannas	71	Treeless
Lake Chad flooded savanna	Flooded Grasslands & Savannas	72	Treeless
Makgadikgadi halophytics	Flooded Grasslands & Savannas	73	Treeless
Nenjiang River grassland	Flooded Grasslands & Savannas	743	Treeless
Nile Delta flooded savanna	Flooded Grasslands & Savannas	744	Treeless
Orinoco wetlands	Flooded Grasslands & Savannas	583	Mosaic
Pantanal	Flooded Grasslands & Savannas	584	Mosaic
Paraná flooded savanna	Flooded Grasslands & Savannas	585	Treeless
Rann of Kutch seasonal salt marsh	Flooded Grasslands & Savannas	312	Treeless
Saharan halophytics	Flooded Grasslands & Savannas	745	Treeless
Southern Cone Mesopotamian savanna	Flooded Grasslands & Savannas	586	Open

Ecoregion	Biome	ID	Category
Sudd flooded grasslands	Flooded Grasslands & Savannas	74	Treeless
Suiphun-Khanka meadows and forest meadows	Flooded Grasslands & Savannas	746	Mosaic
Tigris-Euphrates alluvial salt marsh	Flooded Grasslands & Savannas	747	Treeless
Yellow Sea saline meadow	Flooded Grasslands & Savannas	748	Treeless
Zambezian coastal flooded savanna	Flooded Grasslands & Savannas	75	Mosaic
Zambezian flooded grasslands	Flooded Grasslands & Savannas	76	Treeless
Amazon-Orinoco-Southern Caribbean mangroves	Mangroves	611	Forest
Bahamian-Antillean mangroves	Mangroves	612	Forest
Central African mangroves	Mangroves	111	Forest
East African mangroves	Mangroves	112	Forest
Guinean mangroves	Mangroves	113	Forest
Indochina mangroves	Mangroves	319	Forest
Indus River Delta-Arabian Sea mangroves	Mangroves	320	Forest
Madagascar mangroves	Mangroves	114	Forest
Mesoamerican Gulf-Caribbean mangroves	Mangroves	613	Forest
Myanmar Coast mangroves	Mangroves	321	Forest
New Guinea mangroves	Mangroves	217	Forest
Northern Mesoamerican Pacific mangroves	Mangroves	614	Forest
Red Sea mangroves	Mangroves	115	Forest
South American Pacific mangroves	Mangroves	615	Forest
Southern Africa mangroves	Mangroves	116	Forest
Southern Atlantic Brazilian mangroves	Mangroves	616	Forest
Southern Mesoamerican Pacific mangroves	Mangroves	617	Forest
Sunda Shelf mangroves	Mangroves	322	Forest
Sundarbans mangroves	Mangroves	323	Forest
Aegean and Western Turkey sclerophyllous and mixed forests	Mediterranean Forests, Woodlands & Scrub	785	Forest
Albany thickets	Mediterranean Forests, Woodlands & Scrub	88	Treeless
Anatolian conifer and deciduous mixed forests	Mediterranean Forests, Woodlands & Scrub	786	Forest
California coastal sage and chaparral	Mediterranean Forests, Woodlands & Scrub	422	Mosaic
California interior chaparral and woodlands	Mediterranean Forests, Woodlands & Scrub	423	Mosaic
California montane chaparral and woodlands	Mediterranean Forests, Woodlands & Scrub	424	Mosaic
Canary Islands dry woodlands and forests	Mediterranean Forests, Woodlands & Scrub	787	Forest
Chilean Matorral	Mediterranean Forests, Woodlands & Scrub	596	Mosaic
Coolgardie woodlands	Mediterranean Forests, Woodlands & Scrub	197	Open
Corsican montane broadleaf and mixed forests	Mediterranean Forests, Woodlands & Scrub	788	Forest

Ecoregion	Biome	ID	Category
Crete Mediterranean forests	Mediterranean Forests, Woodlands & Scrub	789	Forest
Cyprus Mediterranean forests	Mediterranean Forests, Woodlands & Scrub	790	Forest
Eastern Mediterranean conifer-broadleaf forests	Mediterranean Forests, Woodlands & Scrub	791	Forest
Esperance mallee	Mediterranean Forests, Woodlands & Scrub	198	Open
Eyre and York mallee	Mediterranean Forests, Woodlands & Scrub	199	Open
Flinders-Lofty montane woodlands	Mediterranean Forests, Woodlands & Scrub	200	Open
Fynbos shrubland	Mediterranean Forests, Woodlands & Scrub	89	Treeless
Hampton mallee and woodlands	Mediterranean Forests, Woodlands & Scrub	201	Open
Iberian conifer forests	Mediterranean Forests, Woodlands & Scrub	792	Forest
Iberian sclerophyllous and semi-deciduous forests	Mediterranean Forests, Woodlands & Scrub	793	Forest
Illyrian deciduous forests	Mediterranean Forests, Woodlands & Scrub	794	Forest
Italian sclerophyllous and semi-deciduous forests	Mediterranean Forests, Woodlands & Scrub	795	Forest
Jarrah-Karri forest and shrublands	Mediterranean Forests, Woodlands & Scrub	202	Forest
Mediterranean Acacia-Argania dry woodlands and succulent thickets	Mediterranean Forests, Woodlands & Scrub	796	Open
Mediterranean dry woodlands and steppe	Mediterranean Forests, Woodlands & Scrub	797	Open
Mediterranean woodlands and forests	Mediterranean Forests, Woodlands & Scrub	798	Forest
Murray-Darling woodlands and mallee	Mediterranean Forests, Woodlands & Scrub	203	Open
Naracoorte woodlands	Mediterranean Forests, Woodlands & Scrub	204	Open
Northeast Spain and Southern France Mediterranean forests	Mediterranean Forests, Woodlands & Scrub	799	Forest
Northwest Iberian montane forests	Mediterranean Forests, Woodlands & Scrub	800	Forest
Pindus Mountains mixed forests	Mediterranean Forests, Woodlands & Scrub	801	Forest
Renosterveld shrubland	Mediterranean Forests, Woodlands & Scrub	90	Treeless
Santa Lucia Montane Chaparral & Woodlands	Mediterranean Forests, Woodlands & Scrub	425	Open
South Apennine mixed montane forests	Mediterranean Forests, Woodlands & Scrub	802	Forest
Southeast Iberian shrubs and woodlands	Mediterranean Forests, Woodlands & Scrub	803	Open

Ecoregion	Biome	ID	Category
Southern Anatolian montane conifer and deciduous forests	Mediterranean Forests, Woodlands & Scrub	804	Forest
Southwest Australia savanna	Mediterranean Forests, Woodlands & Scrub	205	Open
Southwest Australia woodlands	Mediterranean Forests, Woodlands & Scrub	206	Open
Southwest Iberian Mediterranean sclerophyllous and mixed forests	Mediterranean Forests, Woodlands & Scrub	805	Forest
Tyrrhenian-Adriatic sclerophyllous and mixed forests	Mediterranean Forests, Woodlands & Scrub	806	Forest
Altai alpine meadow and tundra	Montane Grasslands & Shrublands	749	Treeless
Angolan montane forest-grassland	Montane Grasslands & Shrublands	77	Mosaic
Australian Alps montane grasslands	Montane Grasslands & Shrublands	193	Treeless
Central Andean dry puna	Montane Grasslands & Shrublands	587	Treeless
Central Andean puna	Montane Grasslands & Shrublands	588	Treeless
Central Andean wet puna	Montane Grasslands & Shrublands	589	Treeless
Papuan Central Range sub-alpine grasslands	Montane Grasslands & Shrublands	195	Treeless
Central Tibetan Plateau alpine steppe	Montane Grasslands & Shrublands	750	Treeless
Cordillera Central páramo	Montane Grasslands & Shrublands	590	Treeless
Cordillera de Merida páramo	Montane Grasslands & Shrublands	591	Treeless
Eastern Himalayan alpine shrub and meadows	Montane Grasslands & Shrublands	751	Treeless
Ethiopian montane grasslands and woodlands	Montane Grasslands & Shrublands	79	Mosaic
Ethiopian montane moorlands	Montane Grasslands & Shrublands	80	Treeless
Ghorat-Hazarajat alpine meadow	Montane Grasslands & Shrublands	752	Treeless
High Monte	Montane Grasslands & Shrublands	592	Treeless
Highveld grasslands	Montane Grasslands & Shrublands	81	Treeless
Hindu Kush alpine meadow	Montane Grasslands & Shrublands	753	Treeless
Jos Plateau forest-grassland	Montane Grasslands & Shrublands	82	Mosaic
Karakoram-West Tibetan Plateau alpine steppe	Montane Grasslands & Shrublands	754	Treeless
Khangai Mountains alpine meadow	Montane Grasslands & Shrublands	755	Treeless
Kopet Dag woodlands and forest steppe	Montane Grasslands & Shrublands	756	Treeless
Kuh Rud and Eastern Iran montane woodlands	Montane Grasslands & Shrublands	757	Treeless
Madagascar ericoid thickets	Montane Grasslands & Shrublands	83	Treeless
Mediterranean High Atlas juniper steppe	Montane Grasslands & Shrublands	758	Treeless
Mulanje Montane forest-grassland	Montane Grasslands & Shrublands	84	Mosaic
North Tibetan Plateau-Kunlun Mountains alpine desert	Montane Grasslands & Shrublands	759	Treeless
Northern Andean páramo	Montane Grasslands & Shrublands	593	Treeless
Northwestern Himalayan alpine shrub and meadows	Montane Grasslands & Shrublands	760	Treeless
Nyanga-Chimanimani Montane forest-grassland	Montane Grasslands & Shrublands	85	Mosaic
Ordos Plateau steppe	Montane Grasslands & Shrublands	761	Treeless
Pamir alpine desert and tundra	Montane Grasslands & Shrublands	762	Treeless

Ecoregion	Biome	ID	Category
Qilian Mountains subalpine meadows	Montane Grasslands & Shrublands	763	Treeless
Rwenzori-Virunga montane moorlands	Montane Grasslands & Shrublands	86	Treeless
Santa Marta páramo	Montane Grasslands & Shrublands	594	Treeless
Sayan alpine meadows and tundra	Montane Grasslands & Shrublands	764	Treeless
New Zealand South Island montane grasslands	Montane Grasslands & Shrublands	194	Treeless
Southeast Tibet shrublands and meadows	Montane Grasslands & Shrublands	765	Treeless
Southern Andean steppe	Montane Grasslands & Shrublands	595	Treeless
Southern Rift Montane forest-grassland	Montane Grasslands & Shrublands	87	Mosaic
Sulaiman Range alpine meadows	Montane Grasslands & Shrublands	766	Treeless
Tian Shan montane steppe and meadows	Montane Grasslands & Shrublands	767	Treeless
Tibetan Plateau alpine shrublands and meadows	Montane Grasslands & Shrublands	768	Treeless
Western Himalayan alpine shrub and meadows	Montane Grasslands & Shrublands	769	Treeless
Yarlung Zanbo arid steppe	Montane Grasslands & Shrublands	770	Treeless
East African montane moorlands	Montane Grasslands & Shrublands	78	Treeless
Kinabalu montane alpine meadows	Montane Grasslands & Shrublands	313	Treeless
Rock and Ice	N/A	0	N/A
Allegheny Highlands forests	Temperate Broadleaf & Mixed Forests	328	Forest
Appalachian-Blue Ridge forests	Temperate Broadleaf & Mixed Forests	331	Forest
Appalachian mixed mesophytic forests	Temperate Broadleaf & Mixed Forests	329	Forest
Appalachian Piedmont forests	Temperate Broadleaf & Mixed Forests	330	Forest
Appenine deciduous montane forests	Temperate Broadleaf & Mixed Forests	644	Forest
European Atlantic mixed forests	Temperate Broadleaf & Mixed Forests	664	Forest
Azores temperate mixed forests	Temperate Broadleaf & Mixed Forests	645	Forest
Balkan mixed forests	Temperate Broadleaf & Mixed Forests	646	Forest
Baltic mixed forests	Temperate Broadleaf & Mixed Forests	647	Forest
Cantabrian mixed forests	Temperate Broadleaf & Mixed Forests	648	Forest
Caspian Hyrcanian mixed forests	Temperate Broadleaf & Mixed Forests	649	Forest
Caucasus mixed forests	Temperate Broadleaf & Mixed Forests	650	Forest
Celtic broadleaf forests	Temperate Broadleaf & Mixed Forests	651	Forest
Central Anatolian steppe and woodlands	Temperate Broadleaf & Mixed Forests	652	Open

Ecoregion	Biome	ID	Category
Central China Loess Plateau mixed forests	Temperate Broadleaf & Mixed Forests	653	Forest
Central European mixed forests	Temperate Broadleaf & Mixed Forests	654	Forest
Central Korean deciduous forests	Temperate Broadleaf & Mixed Forests	655	Forest
Changbai Mountains mixed forests	Temperate Broadleaf & Mixed Forests	656	Forest
Changjiang Plain evergreen forests	Temperate Broadleaf & Mixed Forests	657	Forest
Chatham Island temperate forests	Temperate Broadleaf & Mixed Forests	167	Forest
Crimean Submediterranean forest complex	Temperate Broadleaf & Mixed Forests	658	Forest
Daba Mountains evergreen forests	Temperate Broadleaf & Mixed Forests	659	Forest
Dinaric Mountains mixed forests	Temperate Broadleaf & Mixed Forests	660	Forest
East Central Texas forests	Temperate Broadleaf & Mixed Forests	332	Forest
East European forest steppe	Temperate Broadleaf & Mixed Forests	661	Mosaic
Eastern Anatolian deciduous forests	Temperate Broadleaf & Mixed Forests	662	Forest
Eastern Australian temperate forests	Temperate Broadleaf & Mixed Forests	168	Forest
Eastern Canadian Forest-Boreal transition	Temperate Broadleaf & Mixed Forests	333	Forest
Eastern Great Lakes lowland forests	Temperate Broadleaf & Mixed Forests	334	Forest
Eastern Himalayan broadleaf forests	Temperate Broadleaf & Mixed Forests	306	Forest
English Lowlands beech forests	Temperate Broadleaf & Mixed Forests	663	Forest
Euxine-Colchic broadleaf forests	Temperate Broadleaf & Mixed Forests	665	Forest
Fiordland temperate forests	Temperate Broadleaf & Mixed Forests	169	Forest
Gulf of St. Lawrence lowland forests	Temperate Broadleaf & Mixed Forests	335	Forest
Hokkaido deciduous forests	Temperate Broadleaf & Mixed Forests	666	Forest
Huang He Plain mixed forests	Temperate Broadleaf & Mixed Forests	667	Forest
Interior Plateau US Hardwood Forests	Temperate Broadleaf & Mixed Forests	336	Forest
Juan Fernández Islands temperate forests	Temperate Broadleaf & Mixed Forests	560	Forest
Madeira evergreen forests	Temperate Broadleaf & Mixed Forests	668	Forest

Ecoregion	Biome	ID	Category
Magellanic subpolar forests	Temperate Broadleaf & Mixed Forests	561	Forest
Manchurian mixed forests	Temperate Broadleaf & Mixed Forests	669	Forest
Mississippi lowland forests	Temperate Broadleaf & Mixed Forests	337	Forest
Nelson Coast temperate forests	Temperate Broadleaf & Mixed Forests	170	Forest
New England-Acadian forests	Temperate Broadleaf & Mixed Forests	338	Forest
Nihonkai evergreen forests	Temperate Broadleaf & Mixed Forests	670	Forest
Nihonkai montane deciduous forests	Temperate Broadleaf & Mixed Forests	671	Forest
New Zealand North Island temperate forests	Temperate Broadleaf & Mixed Forests	171	Forest
Northeast China Plain deciduous forests	Temperate Broadleaf & Mixed Forests	673	Forest
Northeast US Coastal forests	Temperate Broadleaf & Mixed Forests	339	Forest
Northern Triangle temperate forests	Temperate Broadleaf & Mixed Forests	307	Forest
Northland temperate kauri forests	Temperate Broadleaf & Mixed Forests	173	Forest
Ozark Highlands mixed forests	Temperate Broadleaf & Mixed Forests	340	Forest
Ozark Mountain forests	Temperate Broadleaf & Mixed Forests	341	Forest
Pannonian mixed forests	Temperate Broadleaf & Mixed Forests	674	Forest
Po Basin mixed forests	Temperate Broadleaf & Mixed Forests	675	Forest
Pyrenees conifer and mixed forests	Temperate Broadleaf & Mixed Forests	676	Forest
Qin Ling Mountains deciduous forests	Temperate Broadleaf & Mixed Forests	677	Forest
Rakiura Island temperate forests	Temperate Broadleaf & Mixed Forests	174	Forest
Richmond temperate forests	Temperate Broadleaf & Mixed Forests	175	Forest
Rodope montane mixed forests	Temperate Broadleaf & Mixed Forests	678	Forest
San Félix-San Ambrosio Islands temperate forests	Temperate Broadleaf & Mixed Forests	562	Forest
Sarmatic mixed forests	Temperate Broadleaf & Mixed Forests	679	Forest
Sichuan Basin evergreen broadleaf forests	Temperate Broadleaf & Mixed Forests	680	Forest
New Zealand South Island temperate forests	Temperate Broadleaf & Mixed Forests	172	Forest

Ecoregion	Biome	ID	Category
Southeast Australia temperate forests	Temperate Broadleaf & Mixed Forests	176	Forest
Southern Great Lakes forests	Temperate Broadleaf & Mixed Forests	342	Forest
Southern Korea evergreen forests	Temperate Broadleaf & Mixed Forests	681	Forest
Taiheiyo evergreen forests	Temperate Broadleaf & Mixed Forests	682	Forest
Taiheiyo montane deciduous forests	Temperate Broadleaf & Mixed Forests	683	Forest
Tarim Basin deciduous forests and steppe	Temperate Broadleaf & Mixed Forests	684	Mosaic
Tasmanian Central Highland forests	Temperate Broadleaf & Mixed Forests	177	Forest
Tasmanian temperate forests	Temperate Broadleaf & Mixed Forests	178	Forest
Tasmanian temperate rain forests	Temperate Broadleaf & Mixed Forests	179	Forest
Upper Midwest US forest-savanna transition	Temperate Broadleaf & Mixed Forests	343	Mosaic
Ussuri broadleaf and mixed forests	Temperate Broadleaf & Mixed Forests	685	Forest
Valdivian temperate forests	Temperate Broadleaf & Mixed Forests	563	Forest
Western European broadleaf forests	Temperate Broadleaf & Mixed Forests	686	Forest
Western Great Lakes forests	Temperate Broadleaf & Mixed Forests	344	Forest
Western Himalayan broadleaf forests	Temperate Broadleaf & Mixed Forests	308	Forest
Western Siberian hemiboreal forests	Temperate Broadleaf & Mixed Forests	687	Forest
Westland temperate forests	Temperate Broadleaf & Mixed Forests	180	Forest
Zagros Mountains forest steppe	Temperate Broadleaf & Mixed Forests	688	Mosaic
North Atlantic moist mixed forests	Temperate Broadleaf & Mixed Forests	672	Forest
Alberta-British Columbia foothills forests	Temperate Conifer Forests	345	Forest
Alps conifer and mixed forests	Temperate Conifer Forests	689	Forest
Altai montane forest and forest steppe	Temperate Conifer Forests	690	Mosaic
Arizona Mountains forests	Temperate Conifer Forests	346	Forest
Atlantic coastal pine barrens	Temperate Conifer Forests	347	Mosaic
Blue Mountains forests	Temperate Conifer Forests	348	Forest
British Columbia coastal conifer forests	Temperate Conifer Forests	349	Forest
Caledon conifer forests	Temperate Conifer Forests	691	Forest
Carpathian montane forests	Temperate Conifer Forests	692	Forest
Central-Southern Cascades Forests	Temperate Conifer Forests	352	Forest

Ecoregion	Biome	ID	Category
Central British Columbia Mountain forests	Temperate Conifer Forests	350	Forest
Central Pacific Northwest coastal forests	Temperate Conifer Forests	351	Forest
Colorado Rockies forests	Temperate Conifer Forests	353	Forest
Da Hinggan-Dzhagdy Mountains conifer forests	Temperate Conifer Forests	693	Forest
East Afghan montane conifer forests	Temperate Conifer Forests	694	Forest
Eastern Cascades forests	Temperate Conifer Forests	354	Forest
Eastern Himalayan subalpine conifer forests	Temperate Conifer Forests	309	Forest
Elburz Range forest steppe	Temperate Conifer Forests	695	Mosaic
Fraser Plateau and Basin conifer forests	Temperate Conifer Forests	355	Forest
Great Basin montane forests	Temperate Conifer Forests	356	Forest
Helanshan montane conifer forests	Temperate Conifer Forests	696	Forest
Hengduan Mountains subalpine conifer forests	Temperate Conifer Forests	697	Forest
Hokkaido montane conifer forests	Temperate Conifer Forests	698	Forest
Honshu alpine conifer forests	Temperate Conifer Forests	699	Forest
Khangai Mountains conifer forests	Temperate Conifer Forests	700	Forest
Klamath-Siskiyou forests	Temperate Conifer Forests	357	Forest
Mediterranean conifer and mixed forests	Temperate Conifer Forests	701	Forest
Northeast Himalayan subalpine conifer forests	Temperate Conifer Forests	702	Forest
Northern Anatolian conifer and deciduous forests	Temperate Conifer Forests	703	Forest
Northern California coastal forests	Temperate Conifer Forests	359	Forest
Nujiang Langcang Gorge alpine conifer and mixed forests	Temperate Conifer Forests	704	Forest
Okanogan dry forests	Temperate Conifer Forests	362	Mosaic
Piney Woods	Temperate Conifer Forests	363	Mosaic
Puget lowland forests	Temperate Conifer Forests	364	Forest
Qilian Mountains conifer forests	Temperate Conifer Forests	705	Forest
Qionglai-Minshan conifer forests	Temperate Conifer Forests	706	Forest
Sayan montane conifer forests	Temperate Conifer Forests	707	Forest
Scandinavian coastal conifer forests	Temperate Conifer Forests	708	Forest
Sierra Nevada forests	Temperate Conifer Forests	366	Forest
South Central Rockies forests	Temperate Conifer Forests	367	Forest
Tian Shan montane conifer forests	Temperate Conifer Forests	709	Forest
Wasatch and Uinta montane forests	Temperate Conifer Forests	368	Forest
Western Himalayan subalpine conifer forests	Temperate Conifer Forests	310	Forest
Queen Charlotte Islands conifer forests	Temperate Conifer Forests	365	Forest
Northern Pacific Alaskan coastal forests	Temperate Conifer Forests	360	Forest
North Cascades conifer forests	Temperate Conifer Forests	358	Forest
Northern Rockies conifer forests	Temperate Conifer Forests	361	Forest
Al-Hajar foothill xeric woodlands and shrublands	Temperate Grasslands, Savannas & Shrublands	722	Open
Al-Hajar montane woodlands and shrublands	Temperate Grasslands, Savannas & Shrublands	723	Open

Ecoregion	Biome	ID	Category
Alai-Western Tian Shan steppe	Temperate Grasslands, Savannas & Shrublands	721	Treeless
Altai steppe and semi-desert	Temperate Grasslands, Savannas & Shrublands	724	Treeless
Amsterdam-Saint Paul Islands temperate grasslands	Temperate Grasslands, Savannas & Shrublands	67	Treeless
California Central Valley grasslands	Temperate Grasslands, Savannas & Shrublands	385	Treeless
Canadian Aspen forests and parklands	Temperate Grasslands, Savannas & Shrublands	386	Mosaic
Canterbury-Otago tussock grasslands	Temperate Grasslands, Savannas & Shrublands	190	Treeless
Central Anatolian steppe	Temperate Grasslands, Savannas & Shrublands	725	Treeless
Central-Southern US mixed grasslands	Temperate Grasslands, Savannas & Shrublands	389	Treeless
Central US forest-grasslands transition	Temperate Grasslands, Savannas & Shrublands	387	Mosaic
Central Tallgrass prairie	Temperate Grasslands, Savannas & Shrublands	388	Treeless
Cross-Timbers savanna-woodland	Temperate Grasslands, Savannas & Shrublands	390	Mosaic
Daurian forest steppe	Temperate Grasslands, Savannas & Shrublands	726	Treeless
Eastern Anatolian montane steppe	Temperate Grasslands, Savannas & Shrublands	727	Treeless
Eastern Australia mulga shrublands	Temperate Grasslands, Savannas & Shrublands	191	Treeless
Edwards Plateau savanna	Temperate Grasslands, Savannas & Shrublands	391	Open
Emin Valley steppe	Temperate Grasslands, Savannas & Shrublands	728	Treeless
Espinal	Temperate Grasslands, Savannas & Shrublands	575	Open
Faroe Islands boreal grasslands	Temperate Grasslands, Savannas & Shrublands	729	Treeless
Flint Hills tallgrass prairie	Temperate Grasslands, Savannas & Shrublands	392	Treeless
Gissaro-Alai open woodlands	Temperate Grasslands, Savannas & Shrublands	730	Open
Humid Pampas	Temperate Grasslands, Savannas & Shrublands	576	Treeless
Kazakh forest steppe	Temperate Grasslands, Savannas & Shrublands	731	Treeless
Kazakh steppe	Temperate Grasslands, Savannas & Shrublands	732	Treeless
Kazakh upland steppe	Temperate Grasslands, Savannas & Shrublands	733	Treeless
Low Monte	Temperate Grasslands, Savannas & Shrublands	577	Treeless

Ecoregion	Biome	ID	Category
Mid-Atlantic US coastal savannas	Temperate Grasslands, Savannas & Shrublands	393	Open
Mongolian-Manchurian grassland	Temperate Grasslands, Savannas & Shrublands	734	Treeless
Montana Valley and Foothill grasslands	Temperate Grasslands, Savannas & Shrublands	394	Treeless
Nebraska Sand Hills mixed grasslands	Temperate Grasslands, Savannas & Shrublands	395	Treeless
Northern Shortgrass prairie	Temperate Grasslands, Savannas & Shrublands	396	Treeless
Northern Tallgrass prairie	Temperate Grasslands, Savannas & Shrublands	397	Treeless
Palouse prairie	Temperate Grasslands, Savannas & Shrublands	398	Treeless
Patagonian steppe	Temperate Grasslands, Savannas & Shrublands	578	Treeless
Pontic steppe	Temperate Grasslands, Savannas & Shrublands	735	Treeless
Sayan Intermontane steppe	Temperate Grasslands, Savannas & Shrublands	736	Treeless
Selenge-Orkhon forest steppe	Temperate Grasslands, Savannas & Shrublands	737	Mosaic
South Siberian forest steppe	Temperate Grasslands, Savannas & Shrublands	738	Mosaic
Southeast Australia temperate savanna	Temperate Grasslands, Savannas & Shrublands	192	Open
Southeast US mixed woodlands and savannas	Temperate Grasslands, Savannas & Shrublands	400	Open
Southeast US conifer savannas	Temperate Grasslands, Savannas & Shrublands	399	Open
Syrian xeric grasslands and shrublands	Temperate Grasslands, Savannas & Shrublands	739	Treeless
Texas blackland prairies	Temperate Grasslands, Savannas & Shrublands	401	Treeless
Tian Shan foothill arid steppe	Temperate Grasslands, Savannas & Shrublands	740	Treeless
Tristan Da Cunha-Gough Islands shrub and grasslands	Temperate Grasslands, Savannas & Shrublands	68	Treeless
Western shortgrass prairie	Temperate Grasslands, Savannas & Shrublands	402	Treeless
Willamette Valley oak savanna	Temperate Grasslands, Savannas & Shrublands	403	Open
Bermuda subtropical conifer forests	Tropical & Subtropical Coniferous Forests	325	Forest
Central American pine-oak forests	Tropical & Subtropical Coniferous Forests	553	Forest
Cuban pine forests	Tropical & Subtropical Coniferous Forests	554	Forest
Himalayan subtropical pine forests	Tropical & Subtropical Coniferous Forests	302	Forest

Ecoregion	Biome	ID	Category
Hispaniolan pine forests	Tropical & Subtropical Coniferous Forests	555	Forest
Luzon tropical pine forests	Tropical & Subtropical Coniferous Forests	303	Forest
Northeast India-Myanmar pine forests	Tropical & Subtropical Coniferous Forests	304	Forest
Sierra de la Laguna pine-oak forests	Tropical & Subtropical Coniferous Forests	556	Forest
Sierra Madre de Oaxaca pine-oak forests	Tropical & Subtropical Coniferous Forests	557	Forest
Sierra Madre del Sur pine-oak forests	Tropical & Subtropical Coniferous Forests	558	Forest
Sierra Madre Occidental pine-oak forests	Tropical & Subtropical Coniferous Forests	326	Forest
Sierra Madre Oriental pine-oak forests	Tropical & Subtropical Coniferous Forests	327	Forest
Sumatran tropical pine forests	Tropical & Subtropical Coniferous Forests	305	Forest
Trans-Mexican Volcanic Belt pine-oak forests	Tropical & Subtropical Coniferous Forests	559	Forest
Bahamian pineyards	Tropical & Subtropical Coniferous Forests	552	Forest
Apure-Villavicencio dry forests	Tropical & Subtropical Dry Broadleaf Forests	520	Forest
Bajío dry forests	Tropical & Subtropical Dry Broadleaf Forests	521	Forest
Balsas dry forests	Tropical & Subtropical Dry Broadleaf Forests	522	Forest
Bolivian montane dry forests	Tropical & Subtropical Dry Broadleaf Forests	523	Forest
Caatinga	Tropical & Subtropical Dry Broadleaf Forests	525	Mosaic
Cape Verde Islands dry forests	Tropical & Subtropical Dry Broadleaf Forests	31	Forest
Cauca Valley dry forests	Tropical & Subtropical Dry Broadleaf Forests	526	Forest
Central American dry forests	Tropical & Subtropical Dry Broadleaf Forests	527	Forest
Central Deccan Plateau dry deciduous forests	Tropical & Subtropical Dry Broadleaf Forests	290	Forest
Central Indochina dry forests	Tropical & Subtropical Dry Broadleaf Forests	291	Forest
Chhota-Nagpur dry deciduous forests	Tropical & Subtropical Dry Broadleaf Forests	292	Forest
Chiapas Depression dry forests	Tropical & Subtropical Dry Broadleaf Forests	528	Forest
Chiquitano dry forests	Tropical & Subtropical Dry Broadleaf Forests	529	Forest
Cuban dry forests	Tropical & Subtropical Dry Broadleaf Forests	530	Forest

Ecoregion	Biome	ID	Category
East Deccan dry-evergreen forests	Tropical & Subtropical Dry Broadleaf Forests	293	Forest
Ecuadorian dry forests	Tropical & Subtropical Dry Broadleaf Forests	531	Forest
Fiji tropical dry forests	Tropical & Subtropical Dry Broadleaf Forests	635	Forest
Hawai'i tropical dry forests	Tropical & Subtropical Dry Broadleaf Forests	636	Forest
Hispaniolan dry forests	Tropical & Subtropical Dry Broadleaf Forests	532	Forest
Irrawaddy dry forests	Tropical & Subtropical Dry Broadleaf Forests	294	Forest
Islas Revillagigedo dry forests	Tropical & Subtropical Dry Broadleaf Forests	533	Forest
Jalisco dry forests	Tropical & Subtropical Dry Broadleaf Forests	534	Forest
Jamaican dry forests	Tropical & Subtropical Dry Broadleaf Forests	535	Forest
Khathiar-Gir dry deciduous forests	Tropical & Subtropical Dry Broadleaf Forests	295	Forest
Lara–Falcón dry forests	Tropical & Subtropical Dry Broadleaf Forests	536	Forest
Lesser Sundas deciduous forests	Tropical & Subtropical Dry Broadleaf Forests	163	Forest
Madagascar dry deciduous forests	Tropical & Subtropical Dry Broadleaf Forests	32	Forest
Magdalena Valley dry forests	Tropical & Subtropical Dry Broadleaf Forests	538	Forest
Maracaibo dry forests	Tropical & Subtropical Dry Broadleaf Forests	539	Forest
Maranhão Babaçu forests	Tropical & Subtropical Dry Broadleaf Forests	540	Forest
Marianas tropical dry forests	Tropical & Subtropical Dry Broadleaf Forests	637	Forest
Narmada Valley dry deciduous forests	Tropical & Subtropical Dry Broadleaf Forests	296	Forest
New Caledonia dry forests	Tropical & Subtropical Dry Broadleaf Forests	164	Forest
North Deccan dry deciduous forests	Tropical & Subtropical Dry Broadleaf Forests	297	Forest
Panamanian dry forests	Tropical & Subtropical Dry Broadleaf Forests	541	Forest
Patía valley dry forests	Tropical & Subtropical Dry Broadleaf Forests	542	Forest
Puerto Rican dry forests	Tropical & Subtropical Dry Broadleaf Forests	543	Forest
Sierra de la Laguna dry forests	Tropical & Subtropical Dry Broadleaf Forests	544	Forest
Sinaloan dry forests	Tropical & Subtropical Dry Broadleaf Forests	545	Forest

Ecoregion	Biome	ID	Category
Sinú Valley dry forests	Tropical & Subtropical Dry Broadleaf Forests	546	Forest
Sonoran-Sinaloan subtropical dry forest	Tropical & Subtropical Dry Broadleaf Forests	324	Forest
South Deccan Plateau dry deciduous forests	Tropical & Subtropical Dry Broadleaf Forests	298	Forest
Southeast Indochina dry evergreen forests	Tropical & Subtropical Dry Broadleaf Forests	299	Forest
Southern Pacific dry forests	Tropical & Subtropical Dry Broadleaf Forests	547	Forest
Southern Vietnam lowland dry forests	Tropical & Subtropical Dry Broadleaf Forests	300	Forest
Sri Lanka dry-zone dry evergreen forests	Tropical & Subtropical Dry Broadleaf Forests	301	Forest
Sumba deciduous forests	Tropical & Subtropical Dry Broadleaf Forests	165	Forest
Timor and Wetar deciduous forests	Tropical & Subtropical Dry Broadleaf Forests	166	Forest
Tumbes-Piura dry forests	Tropical & Subtropical Dry Broadleaf Forests	549	Forest
Veracruz dry forests	Tropical & Subtropical Dry Broadleaf Forests	550	Forest
Yap tropical dry forests	Tropical & Subtropical Dry Broadleaf Forests	638	Forest
Yucatán dry forests	Tropical & Subtropical Dry Broadleaf Forests	551	Forest
Zambezian evergreen dry forests	Tropical & Subtropical Dry Broadleaf Forests	33	Forest
Brazilian Atlantic dry forests	Tropical & Subtropical Dry Broadleaf Forests	524	Forest
Trinidad and Tobago dry forest	Tropical & Subtropical Dry Broadleaf Forests	548	Forest
Lesser Antillean dry forests	Tropical & Subtropical Dry Broadleaf Forests	537	Forest
Angolan mopane woodlands	Tropical & Subtropical Grasslands, Savannas & Shrublands	34	Open
Angolan scarp savanna and woodlands	Tropical & Subtropical Grasslands, Savannas & Shrublands	35	Open
Angolan wet miombo woodlands	Tropical & Subtropical Grasslands, Savannas & Shrublands	36	Open
Arnhem Land tropical savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	181	Open
Ascension scrub and grasslands	Tropical & Subtropical Grasslands, Savannas & Shrublands	37	Treeless
Belizian pine savannas	Tropical & Subtropical Grasslands, Savannas & Shrublands	564	Open
Beni savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	565	Open
Brigalow tropical savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	182	Open

Ecoregion	Biome	ID	Category
Campos Rupestres montane savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	566	Open
Cape York Peninsula tropical savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	183	Open
Carpentaria tropical savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	184	Open
Central bushveld	Tropical & Subtropical Grasslands, Savannas & Shrublands	38	Open
Central Zambezian wet miombo woodlands	Tropical & Subtropical Grasslands, Savannas & Shrublands	39	Open
Cerrado	Tropical & Subtropical Grasslands, Savannas & Shrublands	567	Open
Clipperton Island shrub and grasslands	Tropical & Subtropical Grasslands, Savannas & Shrublands	568	Treeless
Drakensberg Escarpment savanna and thicket	Tropical & Subtropical Grasslands, Savannas & Shrublands	40	Open
Drakensberg grasslands	Tropical & Subtropical Grasslands, Savannas & Shrublands	41	Treeless
Dry Chaco	Tropical & Subtropical Grasslands, Savannas & Shrublands	569	Open
East Sudanian savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	43	Open
Einasleigh upland savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	185	Open
Guianan savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	570	Open
Guinean forest-savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	44	Mosaic
Hawai'i tropical high shrublands	Tropical & Subtropical Grasslands, Savannas & Shrublands	639	Treeless
Hawai'i tropical low shrublands	Tropical & Subtropical Grasslands, Savannas & Shrublands	640	Treeless
Horn of Africa xeric bushlands	Tropical & Subtropical Grasslands, Savannas & Shrublands	45	Treeless
Humid Chaco	Tropical & Subtropical Grasslands, Savannas & Shrublands	571	Mosaic
Itigi-Sumbu thicket	Tropical & Subtropical Grasslands, Savannas & Shrublands	46	Treeless
Kalahari Acacia woodlands	Tropical & Subtropical Grasslands, Savannas & Shrublands	47	Open
Kimberly tropical savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	186	Open
Limpopo lowveld	Tropical & Subtropical Grasslands, Savannas & Shrublands	48	Open
Llanos	Tropical & Subtropical Grasslands, Savannas & Shrublands	572	Open
Mandara Plateau woodlands	Tropical & Subtropical Grasslands, Savannas & Shrublands	49	Open
Masai xeric grasslands and shrublands	Tropical & Subtropical Grasslands, Savannas & Shrublands	50	Treeless

Ecoregion	Biome	ID	Category
Miskito pine forests	Tropical & Subtropical Grasslands, Savannas & Shrublands	573	Open
Mitchell Grass Downs	Tropical & Subtropical Grasslands, Savannas & Shrublands	187	Treeless
Northern Acacia-Commiphora bushlands and thickets	Tropical & Subtropical Grasslands, Savannas & Shrublands	51	Open
Northern Congolian Forest-Savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	52	Mosaic
Northwest Hawai'i scrub	Tropical & Subtropical Grasslands, Savannas & Shrublands	641	Treeless
Sahelian Acacia savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	53	Open
Serengeti volcanic grasslands	Tropical & Subtropical Grasslands, Savannas & Shrublands	54	Treeless
Somali Acacia-Commiphora bushlands and thickets	Tropical & Subtropical Grasslands, Savannas & Shrublands	55	Open
Southern Acacia-Commiphora bushlands and thickets	Tropical & Subtropical Grasslands, Savannas & Shrublands	57	Open
Southern Congolian forest-savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	58	Mosaic
Terai-Duar savanna and grasslands	Tropical & Subtropical Grasslands, Savannas & Shrublands	311	Mosaic
Trans Fly savanna and grasslands	Tropical & Subtropical Grasslands, Savannas & Shrublands	188	Open
Uruguayan savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	574	Open
Victoria Plains tropical savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	189	Open
West Sudanian savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	62	Open
Western Gulf coastal grasslands	Tropical & Subtropical Grasslands, Savannas & Shrublands	384	Treeless
Zambezian-Limpopo mixed woodlands	Tropical & Subtropical Grasslands, Savannas & Shrublands	66	Open
Zambezian Baikiaea woodlands	Tropical & Subtropical Grasslands, Savannas & Shrublands	64	Open
Dry miombo woodlands	Tropical & Subtropical Grasslands, Savannas & Shrublands	42	Open
Zambezian mopane woodlands	Tropical & Subtropical Grasslands, Savannas & Shrublands	65	Open
St. Helena scrub and woodlands	Tropical & Subtropical Grasslands, Savannas & Shrublands	60	Open
Victoria Basin forest-savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	61	Mosaic
Western Congolian forest-savanna	Tropical & Subtropical Grasslands, Savannas & Shrublands	63	Mosaic
Southwest Arabian montane woodlands and grasslands	Tropical & Subtropical Grasslands, Savannas & Shrublands	59	Open
South Arabian fog woodlands, shrublands, and dune	Tropical & Subtropical Grasslands, Savannas & Shrublands	56	Open

Ecoregion	Biome	ID	Category
Admiralty Islands lowland rain forests	Tropical & Subtropical Moist Broadleaf Forests	135	Forest
Albertine Rift montane forests	Tropical & Subtropical Moist Broadleaf Forests	1	Forest
Alto Paraná Atlantic forests	Tropical & Subtropical Moist Broadleaf Forests	439	Forest
Andaman Islands rain forests	Tropical & Subtropical Moist Broadleaf Forests	218	Forest
Araucaria moist forests	Tropical & Subtropical Moist Broadleaf Forests	440	Forest
Atlantic Coast restingas	Tropical & Subtropical Moist Broadleaf Forests	441	Mosaic
Congolian coastal forests	Tropical & Subtropical Moist Broadleaf Forests	5	Forest
Bahia coastal forests	Tropical & Subtropical Moist Broadleaf Forests	442	Forest
Bahia interior forests	Tropical & Subtropical Moist Broadleaf Forests	443	Forest
Banda Sea Islands moist deciduous forests	Tropical & Subtropical Moist Broadleaf Forests	136	Forest
Biak-Numfoor rain forests	Tropical & Subtropical Moist Broadleaf Forests	137	Forest
Bolivian Yungas	Tropical & Subtropical Moist Broadleaf Forests	444	Forest
Borneo lowland rain forests	Tropical & Subtropical Moist Broadleaf Forests	219	Forest
Borneo peat swamp forests	Tropical & Subtropical Moist Broadleaf Forests	221	Forest
Brahmaputra Valley semi-evergreen forests	Tropical & Subtropical Moist Broadleaf Forests	222	Forest
Buru rain forests	Tropical & Subtropical Moist Broadleaf Forests	138	Forest
Caatinga Enclaves moist forests	Tropical & Subtropical Moist Broadleaf Forests	445	Forest
Cameroon Highlands forests	Tropical & Subtropical Moist Broadleaf Forests	2	Forest
Caqueta moist forests	Tropical & Subtropical Moist Broadleaf Forests	446	Forest
Cardamom Mountains rain forests	Tropical & Subtropical Moist Broadleaf Forests	223	Forest
Carolines tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	618	Forest
Catatumbo moist forests	Tropical & Subtropical Moist Broadleaf Forests	447	Forest
Cauca Valley montane forests	Tropical & Subtropical Moist Broadleaf Forests	448	Forest
Cayos Miskitos-San Andrés and Providencia moist forests	Tropical & Subtropical Moist Broadleaf Forests	449	Forest
Central American Atlantic moist forests	Tropical & Subtropical Moist Broadleaf Forests	450	Forest

Ecoregion	Biome	ID	Category
Central American montane forests	Tropical & Subtropical Moist Broadleaf Forests	451	Forest
Central Congolian lowland forests	Tropical & Subtropical Moist Broadleaf Forests	3	Forest
Central Polynesian tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	619	Forest
Central Range Papuan montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	139	Forest
Chao Phraya freshwater swamp forests	Tropical & Subtropical Moist Broadleaf Forests	224	Forest
Chao Phraya lowland moist deciduous forests	Tropical & Subtropical Moist Broadleaf Forests	225	Forest
Chiapas montane forests	Tropical & Subtropical Moist Broadleaf Forests	452	Forest
Chimalapas montane forests	Tropical & Subtropical Moist Broadleaf Forests	453	Forest
Chin Hills-Arakan Yoma montane forests	Tropical & Subtropical Moist Broadleaf Forests	226	Forest
Chocó–Darién moist forests	Tropical & Subtropical Moist Broadleaf Forests	454	Forest
Christmas and Cocos Islands tropical forests	Tropical & Subtropical Moist Broadleaf Forests	227	Forest
Cocos Island moist forests	Tropical & Subtropical Moist Broadleaf Forests	455	Forest
Comoros forests	Tropical & Subtropical Moist Broadleaf Forests	4	Forest
Cook Islands tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	620	Forest
Cordillera La Costa montane forests	Tropical & Subtropical Moist Broadleaf Forests	456	Forest
Cordillera Oriental montane forests	Tropical & Subtropical Moist Broadleaf Forests	457	Forest
Costa Rican seasonal moist forests	Tropical & Subtropical Moist Broadleaf Forests	458	Forest
Cross-Niger transition forests	Tropical & Subtropical Moist Broadleaf Forests	6	Forest
Cross-Sanaga-Bioko coastal forests	Tropical & Subtropical Moist Broadleaf Forests	7	Forest
Cuban moist forests	Tropical & Subtropical Moist Broadleaf Forests	459	Forest
Eastern Congolian swamp forests	Tropical & Subtropical Moist Broadleaf Forests	10	Forest
Eastern Cordillera Real montane forests	Tropical & Subtropical Moist Broadleaf Forests	460	Forest
Eastern Guinean forests	Tropical & Subtropical Moist Broadleaf Forests	11	Forest
East Deccan moist deciduous forests	Tropical & Subtropical Moist Broadleaf Forests	228	Forest
Eastern Java-Bali montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	229	Forest

Ecoregion	Biome	ID	Category
Eastern Java-Bali rain forests	Tropical & Subtropical Moist Broadleaf Forests	230	Forest
Eastern Micronesia tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	621	Forest
Eastern Panamanian montane forests	Tropical & Subtropical Moist Broadleaf Forests	461	Forest
Ethiopian montane forests	Tropical & Subtropical Moist Broadleaf Forests	12	Forest
Fernando de Noronha-Atol das Rocas moist forests	Tropical & Subtropical Moist Broadleaf Forests	462	Forest
Fiji tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	622	Forest
Granitic Seychelles forests	Tropical & Subtropical Moist Broadleaf Forests	13	Forest
Greater Negros-Panay rain forests	Tropical & Subtropical Moist Broadleaf Forests	231	Forest
Guianan freshwater swamp forests	Tropical & Subtropical Moist Broadleaf Forests	463	Forest
Guianan Highlands moist forests	Tropical & Subtropical Moist Broadleaf Forests	464	Forest
Guianan lowland moist forests	Tropical & Subtropical Moist Broadleaf Forests	465	Forest
Guianan piedmont moist forests	Tropical & Subtropical Moist Broadleaf Forests	466	Forest
Guinean montane forests	Tropical & Subtropical Moist Broadleaf Forests	14	Forest
Guizhou Plateau broadleaf and mixed forests	Tropical & Subtropical Moist Broadleaf Forests	642	Forest
Gurupa várzea	Tropical & Subtropical Moist Broadleaf Forests	467	Forest
Hainan Island monsoon rain forests	Tropical & Subtropical Moist Broadleaf Forests	232	Forest
Halmahera rain forests	Tropical & Subtropical Moist Broadleaf Forests	140	Forest
Hawai'i tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	623	Forest
Himalayan subtropical broadleaf forests	Tropical & Subtropical Moist Broadleaf Forests	233	Forest
Hispaniolan moist forests	Tropical & Subtropical Moist Broadleaf Forests	468	Forest
Huon Peninsula montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	141	Forest
Iquitos várzea	Tropical & Subtropical Moist Broadleaf Forests	469	Forest
Irrawaddy freshwater swamp forests	Tropical & Subtropical Moist Broadleaf Forests	234	Forest
Irrawaddy moist deciduous forests	Tropical & Subtropical Moist Broadleaf Forests	235	Forest
Isthmian-Atlantic moist forests	Tropical & Subtropical Moist Broadleaf Forests	470	Forest

Ecoregion	Biome	ID	Category
Isthmian-Pacific moist forests	Tropical & Subtropical Moist Broadleaf Forests	471	Forest
Jamaican moist forests	Tropical & Subtropical Moist Broadleaf Forests	472	Forest
Japurá-Solimões-Negro moist forests	Tropical & Subtropical Moist Broadleaf Forests	473	Forest
Jian Nan subtropical evergreen forests	Tropical & Subtropical Moist Broadleaf Forests	236	Forest
Juruá-Purus moist forests	Tropical & Subtropical Moist Broadleaf Forests	474	Forest
Kayah-Karen montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	237	Forest
Kermadec Islands subtropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	624	Forest
Knysna-Amatole montane forests	Tropical & Subtropical Moist Broadleaf Forests	15	Forest
Kwazulu Natal-Cape coastal forests	Tropical & Subtropical Moist Broadleaf Forests	16	Forest
Leeward Islands moist forests	Tropical & Subtropical Moist Broadleaf Forests	475	Forest
Lord Howe Island subtropical forests	Tropical & Subtropical Moist Broadleaf Forests	142	Forest
Louisiade Archipelago rain forests	Tropical & Subtropical Moist Broadleaf Forests	143	Forest
Lower Gangetic Plains moist deciduous forests	Tropical & Subtropical Moist Broadleaf Forests	238	Forest
Luang Prabang montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	239	Forest
Luzon montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	240	Forest
Luzon rain forests	Tropical & Subtropical Moist Broadleaf Forests	241	Forest
Madagascar humid forests	Tropical & Subtropical Moist Broadleaf Forests	17	Forest
Madagascar subhumid forests	Tropical & Subtropical Moist Broadleaf Forests	18	Forest
Madeira-Tapajós moist forests	Tropical & Subtropical Moist Broadleaf Forests	476	Forest
Magdalena–Urabá moist forests	Tropical & Subtropical Moist Broadleaf Forests	478	Forest
Magdalena Valley montane forests	Tropical & Subtropical Moist Broadleaf Forests	477	Forest
Malabar Coast moist forests	Tropical & Subtropical Moist Broadleaf Forests	242	Forest
Maldives-Lakshadweep-Chagos Archipelago tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	243	Forest
Maputaland coastal forests and woodlands	Tropical & Subtropical Moist Broadleaf Forests	19	Mosaic
Marajó várzea	Tropical & Subtropical Moist Broadleaf Forests	480	Forest

Ecoregion	Biome	ID	Category
Marañón dry forests	Tropical & Subtropical Moist Broadleaf Forests	479	Forest
Marquesas tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	625	Forest
Mascarene forests	Tropical & Subtropical Moist Broadleaf Forests	20	Forest
Mato Grosso tropical dry forests	Tropical & Subtropical Moist Broadleaf Forests	481	Forest
Meghalaya subtropical forests	Tropical & Subtropical Moist Broadleaf Forests	244	Forest
Mentawai Islands rain forests	Tropical & Subtropical Moist Broadleaf Forests	245	Forest
Mindanao-Eastern Visayas rain forests	Tropical & Subtropical Moist Broadleaf Forests	247	Forest
Mindanao montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	246	Forest
Mindoro rain forests	Tropical & Subtropical Moist Broadleaf Forests	248	Forest
Mizoram-Manipur-Kachin rain forests	Tropical & Subtropical Moist Broadleaf Forests	249	Forest
Monte Alegre várzea	Tropical & Subtropical Moist Broadleaf Forests	482	Forest
Mount Cameroon and Bioko montane forests	Tropical & Subtropical Moist Broadleaf Forests	21	Forest
Myanmar coastal rain forests	Tropical & Subtropical Moist Broadleaf Forests	250	Forest
Nansei Islands subtropical evergreen forests	Tropical & Subtropical Moist Broadleaf Forests	251	Forest
Napo moist forests	Tropical & Subtropical Moist Broadleaf Forests	483	Forest
Negro-Branco moist forests	Tropical & Subtropical Moist Broadleaf Forests	484	Forest
New Britain-New Ireland lowland rain forests	Tropical & Subtropical Moist Broadleaf Forests	144	Forest
New Britain-New Ireland montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	145	Forest
New Caledonia rain forests	Tropical & Subtropical Moist Broadleaf Forests	146	Forest
Nicobar Islands rain forests	Tropical & Subtropical Moist Broadleaf Forests	252	Forest
Niger Delta swamp forests	Tropical & Subtropical Moist Broadleaf Forests	22	Forest
Nigerian lowland forests	Tropical & Subtropical Moist Broadleaf Forests	23	Forest
Norfolk Island subtropical forests	Tropical & Subtropical Moist Broadleaf Forests	147	Forest
North Western Ghats moist deciduous forests	Tropical & Subtropical Moist Broadleaf Forests	253	Forest
North Western Ghats montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	254	Forest

Ecoregion	Biome	ID	Category
Northeast Brazil restingas	Tropical & Subtropical Moist Broadleaf Forests	485	Mosaic
Northeast Congolian lowland forests	Tropical & Subtropical Moist Broadleaf Forests	24	Forest
Northern Annamites rain forests	Tropical & Subtropical Moist Broadleaf Forests	255	Forest
Northern Indochina subtropical forests	Tropical & Subtropical Moist Broadleaf Forests	256	Forest
Northern Khorat Plateau moist deciduous forests	Tropical & Subtropical Moist Broadleaf Forests	257	Forest
Northern New Guinea lowland rain and freshwater swamp forests	Tropical & Subtropical Moist Broadleaf Forests	148	Forest
Northern New Guinea montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	149	Forest
Northern Swahili coastal forests	Tropical & Subtropical Moist Broadleaf Forests	25	Forest
Northern Thailand-Laos moist deciduous forests	Tropical & Subtropical Moist Broadleaf Forests	258	Forest
Northern Triangle subtropical forests	Tropical & Subtropical Moist Broadleaf Forests	259	Forest
Northern Vietnam lowland rain forests	Tropical & Subtropical Moist Broadleaf Forests	260	Forest
Northwest Andean montane forests	Tropical & Subtropical Moist Broadleaf Forests	486	Forest
Northwest Congolian lowland forests	Tropical & Subtropical Moist Broadleaf Forests	26	Forest
Oaxacan montane forests	Tropical & Subtropical Moist Broadleaf Forests	487	Forest
Ogasawara subtropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	626	Forest
Orinoco Delta swamp forests	Tropical & Subtropical Moist Broadleaf Forests	488	Forest
Orissa semi-evergreen forests	Tropical & Subtropical Moist Broadleaf Forests	261	Forest
Palau tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	627	Forest
Palawan rain forests	Tropical & Subtropical Moist Broadleaf Forests	262	Forest
Pantanos de Centla	Tropical & Subtropical Moist Broadleaf Forests	489	Mosaic
Pantepui forests & shrublands	Tropical & Subtropical Moist Broadleaf Forests	490	Mosaic
Peninsular Malaysian montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	263	Forest
Peninsular Malaysian peat swamp forests	Tropical & Subtropical Moist Broadleaf Forests	264	Forest
Peninsular Malaysian rain forests	Tropical & Subtropical Moist Broadleaf Forests	265	Forest
Pernambuco coastal forests	Tropical & Subtropical Moist Broadleaf Forests	491	Forest

Ecoregion	Biome	ID	Category
Pernambuco interior forests	Tropical & Subtropical Moist Broadleaf Forests	492	Forest
Peruvian Yungas	Tropical & Subtropical Moist Broadleaf Forests	493	Forest
Petén-Veracruz moist forests	Tropical & Subtropical Moist Broadleaf Forests	494	Forest
Puerto Rican moist forests	Tropical & Subtropical Moist Broadleaf Forests	495	Forest
Purus-Madeira moist forests	Tropical & Subtropical Moist Broadleaf Forests	497	Forest
Purus várzea	Tropical & Subtropical Moist Broadleaf Forests	496	Forest
Queensland tropical rain forests	Tropical & Subtropical Moist Broadleaf Forests	150	Forest
Rapa Nui and Sala y Gómez subtropical forests	Tropical & Subtropical Moist Broadleaf Forests	628	Forest
Red River freshwater swamp forests	Tropical & Subtropical Moist Broadleaf Forests	266	Forest
Rio Negro campinarana	Tropical & Subtropical Moist Broadleaf Forests	498	Mosaic
Samoan tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	629	Forest
Santa Marta montane forests	Tropical & Subtropical Moist Broadleaf Forests	499	Forest
São Tomé, Príncipe, and Annobón forests	Tropical & Subtropical Moist Broadleaf Forests	27	Forest
Seram rain forests	Tropical & Subtropical Moist Broadleaf Forests	151	Forest
Serra do Mar coastal forests	Tropical & Subtropical Moist Broadleaf Forests	500	Forest
Sierra de los Tuxtlas	Tropical & Subtropical Moist Broadleaf Forests	501	Forest
Sierra Madre de Chiapas moist forests	Tropical & Subtropical Moist Broadleaf Forests	502	Forest
Society Islands tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	630	Forest
Solimões-Japurá moist forests	Tropical & Subtropical Moist Broadleaf Forests	503	Forest
Solomon Islands rain forests	Tropical & Subtropical Moist Broadleaf Forests	152	Forest
South China-Vietnam subtropical evergreen forests	Tropical & Subtropical Moist Broadleaf Forests	268	Forest
South China Sea Islands	Tropical & Subtropical Moist Broadleaf Forests	267	Forest
South Taiwan monsoon rain forests	Tropical & Subtropical Moist Broadleaf Forests	269	Forest
South Western Ghats moist deciduous forests	Tropical & Subtropical Moist Broadleaf Forests	270	Forest
South Western Ghats montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	271	Forest

Ecoregion	Biome	ID	Category
Southeast Papuan rain forests	Tropical & Subtropical Moist Broadleaf Forests	153	Forest
Southern Andean Yungas	Tropical & Subtropical Moist Broadleaf Forests	504	Forest
Southern Annamites montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	272	Forest
Southern New Guinea freshwater swamp forests	Tropical & Subtropical Moist Broadleaf Forests	154	Forest
Southern New Guinea lowland rain forests	Tropical & Subtropical Moist Broadleaf Forests	155	Forest
Southern Swahili coastal forests and woodlands	Tropical & Subtropical Moist Broadleaf Forests	28	Mosaic
Southwest Amazon moist forests	Tropical & Subtropical Moist Broadleaf Forests	505	Forest
Southwest Borneo freshwater swamp forests	Tropical & Subtropical Moist Broadleaf Forests	273	Forest
Sri Lanka lowland rain forests	Tropical & Subtropical Moist Broadleaf Forests	274	Forest
Sri Lanka montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	275	Forest
Sulawesi montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	157	Forest
Sulu Archipelago rain forests	Tropical & Subtropical Moist Broadleaf Forests	276	Forest
Sumatran freshwater swamp forests	Tropical & Subtropical Moist Broadleaf Forests	277	Forest
Sumatran lowland rain forests	Tropical & Subtropical Moist Broadleaf Forests	278	Forest
Sumatran montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	279	Forest
Sumatran peat swamp forests	Tropical & Subtropical Moist Broadleaf Forests	280	Forest
Sundaland heath forests	Tropical & Subtropical Moist Broadleaf Forests	281	Forest
Sundarbans freshwater swamp forests	Tropical & Subtropical Moist Broadleaf Forests	282	Forest
Taiwan subtropical evergreen forests	Tropical & Subtropical Moist Broadleaf Forests	283	Forest
Talamancan montane forests	Tropical & Subtropical Moist Broadleaf Forests	506	Forest
Tapajós-Xingu moist forests	Tropical & Subtropical Moist Broadleaf Forests	507	Forest
Tenasserim-South Thailand semi-evergreen rain forests	Tropical & Subtropical Moist Broadleaf Forests	284	Forest
Tongan tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	631	Forest
Tonle Sap-Mekong peat swamp forests	Tropical & Subtropical Moist Broadleaf Forests	286	Forest
Tonle Sap freshwater swamp forests	Tropical & Subtropical Moist Broadleaf Forests	285	Forest

Ecoregion	Biome	ID	Category
Trinidad and Tobago moist forest	Tropical & Subtropical Moist Broadleaf Forests	510	Forest
Trobriand Islands rain forests	Tropical & Subtropical Moist Broadleaf Forests	158	Forest
Tuamotu tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	632	Forest
Tubuai tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	633	Forest
Uatumã-Trombetas moist forests	Tropical & Subtropical Moist Broadleaf Forests	511	Forest
Ucayali moist forests	Tropical & Subtropical Moist Broadleaf Forests	512	Forest
Upper Gangetic Plains moist deciduous forests	Tropical & Subtropical Moist Broadleaf Forests	287	Forest
Vanuatu rain forests	Tropical & Subtropical Moist Broadleaf Forests	159	Forest
Venezuelan Andes montane forests	Tropical & Subtropical Moist Broadleaf Forests	513	Forest
Veracruz moist forests	Tropical & Subtropical Moist Broadleaf Forests	514	Forest
Veracruz montane forests	Tropical & Subtropical Moist Broadleaf Forests	515	Forest
Vogelkop-Aru lowland rain forests	Tropical & Subtropical Moist Broadleaf Forests	161	Forest
Vogelkop montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	160	Forest
Western Congolian swamp forests	Tropical & Subtropical Moist Broadleaf Forests	29	Forest
Western Ecuador moist forests	Tropical & Subtropical Moist Broadleaf Forests	516	Forest
Western Guinean lowland forests	Tropical & Subtropical Moist Broadleaf Forests	30	Forest
Western Java montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	288	Forest
Western Java rain forests	Tropical & Subtropical Moist Broadleaf Forests	289	Forest
Western Polynesian tropical moist forests	Tropical & Subtropical Moist Broadleaf Forests	634	Forest
Windward Islands moist forests	Tropical & Subtropical Moist Broadleaf Forests	517	Forest
Xingu-Tocantins-Araguaia moist forests	Tropical & Subtropical Moist Broadleaf Forests	518	Forest
Yapen rain forests	Tropical & Subtropical Moist Broadleaf Forests	162	Forest
Yucatán moist forests	Tropical & Subtropical Moist Broadleaf Forests	519	Forest
Yunnan Plateau subtropical evergreen forests	Tropical & Subtropical Moist Broadleaf Forests	643	Forest
Tocantins/Pindare moist forests	Tropical & Subtropical Moist Broadleaf Forests	508	Forest

Ecoregion	Biome	ID	Category
Trindade-Martin Vaz Islands tropical forests	Tropical & Subtropical Moist Broadleaf Forests	509	Forest
Sulawesi lowland rain forests	Tropical & Subtropical Moist Broadleaf Forests	156	Forest
East African montane forests	Tropical & Subtropical Moist Broadleaf Forests	8	Forest
Eastern Arc forests	Tropical & Subtropical Moist Broadleaf Forests	9	Forest
Borneo montane rain forests	Tropical & Subtropical Moist Broadleaf Forests	220	Forest
Adelie Land tundra	Tundra	117	Treeless
Ahklun and Kilbuck Upland Tundra	Tundra	404	Treeless
Alaska-St. Elias Range tundra	Tundra	405	Treeless
Aleutian Islands tundra	Tundra	406	Treeless
Antipodes Subantarctic Islands tundra	Tundra	196	Treeless
Arctic coastal tundra	Tundra	407	Treeless
Russian Arctic desert	Tundra	778	Treeless
Arctic foothills tundra	Tundra	408	Treeless
Russian Bering tundra	Tundra	779	Treeless
Beringia lowland tundra	Tundra	409	Treeless
Beringia upland tundra	Tundra	410	Treeless
Brooks-British Range tundra	Tundra	411	Treeless
Canadian Low Arctic tundra	Tundra	413	Treeless
Central South Antarctic Peninsula tundra	Tundra	118	Treeless
Cherskii-Kolyma mountain tundra	Tundra	771	Treeless
Chukchi Peninsula tundra	Tundra	772	Treeless
Davis Highlands tundra	Tundra	415	Treeless
East Antarctic tundra	Tundra	120	Treeless
Ellsworth Land tundra	Tundra	121	Treeless
Ellsworth Mountains tundra	Tundra	122	Treeless
Enderby Land tundra	Tundra	123	Treeless
Canadian High Arctic tundra	Tundra	412	Treeless
Interior Yukon-Alaska alpine tundra	Tundra	416	Treeless
Kalaallit Nunaat High Arctic tundra	Tundra	418	Treeless
Kola Peninsula tundra	Tundra	774	Treeless
Canadian Middle Arctic Tundra	Tundra	414	Treeless
Northeast Antarctic Peninsula tundra	Tundra	126	Treeless
Northwest Antarctic Peninsula tundra	Tundra	127	Treeless
North Victoria Land tundra	Tundra	125	Treeless
Northeast Siberian coastal tundra	Tundra	775	Treeless
Northwest Russian-Novaya Zemlya tundra	Tundra	776	Treeless
Novosibirsk Islands Arctic desert	Tundra	777	Treeless
Ogilvie-MacKenzie alpine tundra	Tundra	419	Treeless
Pacific Coastal Mountain icefields and tundra	Tundra	420	Treeless
Prince Charles Mountains tundra	Tundra	128	Treeless

Ecoregion	Biome	ID	Category
Scandinavian Montane Birch forest and grasslands	Tundra	780	Treeless
Scotia Sea Islands tundra	Tundra	129	Treeless
South Antarctic Peninsula tundra	Tundra	130	Treeless
South Orkney Islands tundra	Tundra	131	Treeless
South Victoria Land tundra	Tundra	132	Treeless
Southern Indian Ocean Islands tundra	Tundra	133	Treeless
Taimyr-Central Siberian tundra	Tundra	781	Treeless
Torngat Mountain tundra	Tundra	421	Treeless
Trans-Baikal Bald Mountain tundra	Tundra	782	Treeless
Transantarctic Mountains tundra	Tundra	134	Treeless
Wrangel Island Arctic desert	Tundra	783	Treeless
Yamal-Gydan tundra	Tundra	784	Treeless
Kamchatka tundra	Tundra	773	Treeless
Kalaallit Nunaat Arctic steppe	Tundra	417	Treeless
Dronning Maud Land tundra	Tundra	119	Treeless
Marie Byrd Land tundra	Tundra	124	Treeless

Table S 5: Sensitivity analysis of thresholds of predicted natural tree cover. *Analyses of site counts were conducted on 687,000 planting sites, while area-based analyses were on 491,069 sites after removal of nested subset polygons to avoid double counting. All numbers are rounded to two decimals.*

Threshold	Number of sites above the threshold	Percentage of sites above the threshold	Area above the threshold (km^2)	Percentage of area above the threshold
0.1	483,898	98.54	5,773,960	96.21
0.2	479,762	97.70	5,565,410	92.74
0.3	423,622	86.27	4,340,190	72.32
0.4	307,362	62.59	3,289,590	54.81
0.5	245,771	50.05	2,559,160	42.64
0.6	229,351	46.70	1,978,470	32.97
0.7	146,833	29.90	1,405,400	23.42
0.8	65,939	13.43	789,847	13.16
0.9	23,095	4.70	355,603	5.93

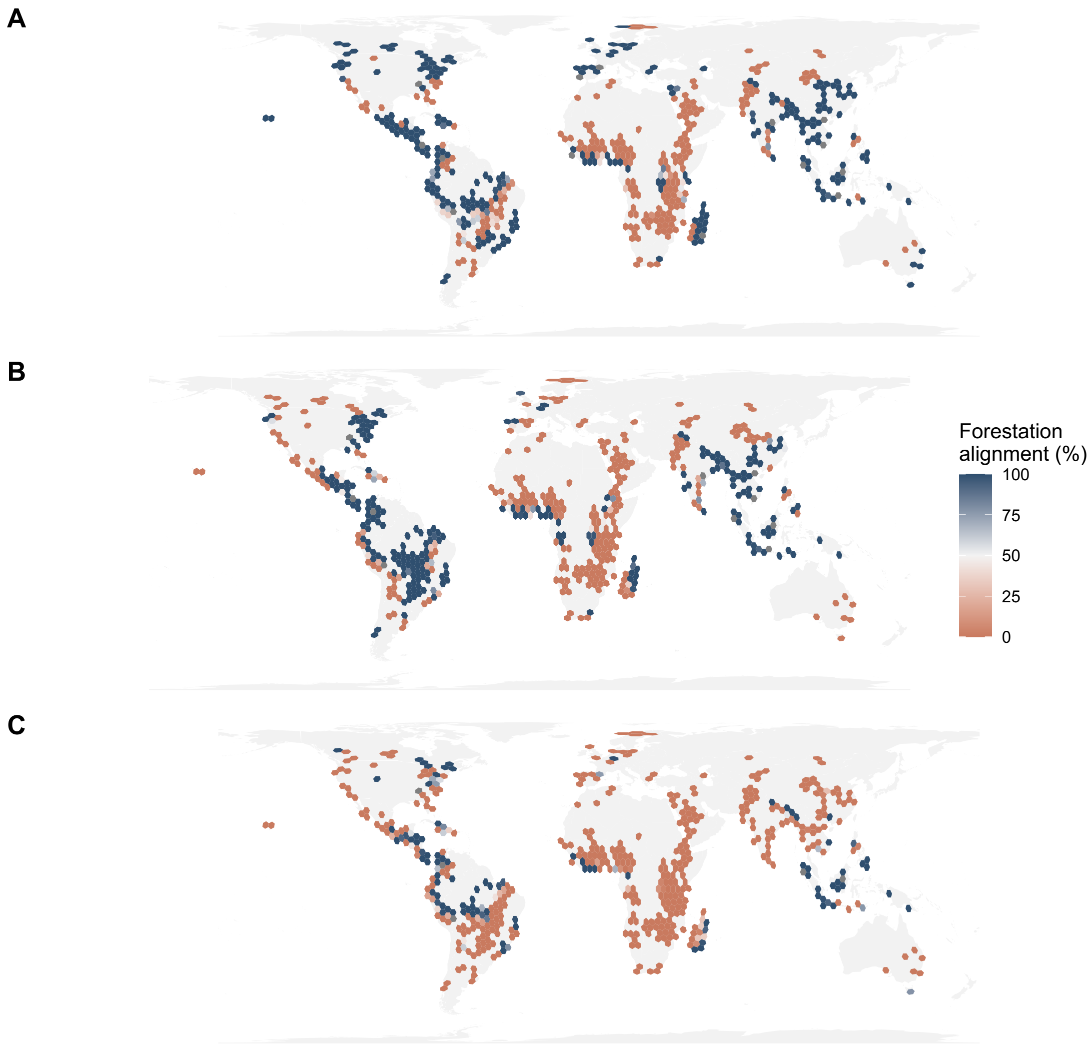

Figure S 1: Global distribution of forestation planting sites classified as correct placed (blue) or incorrectly placed (orange) with forest-dominated ecosystems under three frameworks*. Panels show placement assessed using (A) forested ecoregions, (B) biophysical forest potential, and (C) predicted natural tree cover. Only hexagons with planting sites are shown; grey areas indicate that no projects are present. Differences between panels reflect the distinct ecological criteria used by each of the three frameworks.*

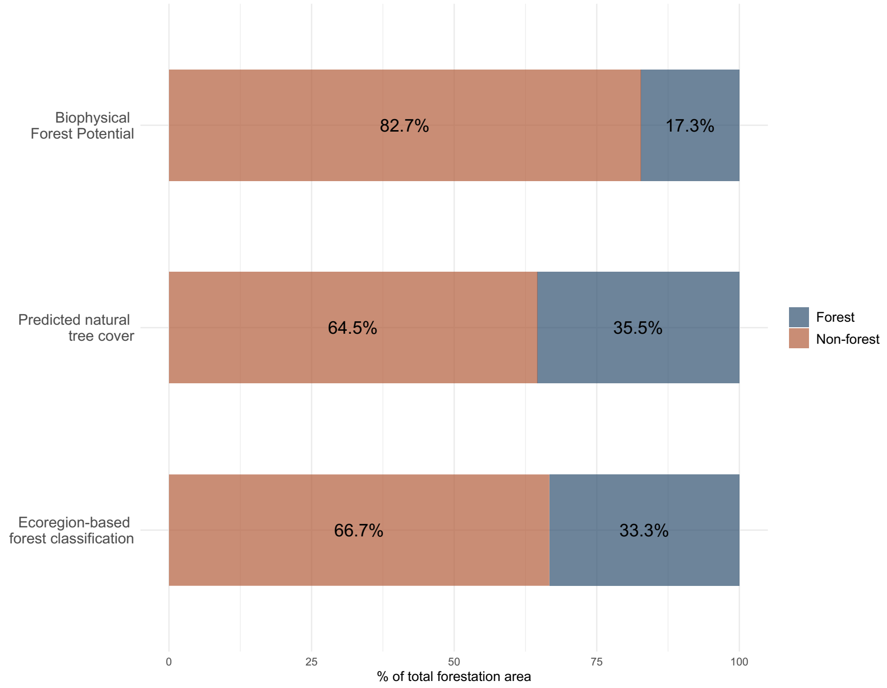

Figure S 2: Global percentage of forestation area located inside and outside forest-dominated systems, by framework (biophysical forest potential, predicted natural tree cover, and ecoregion-based forest classification). *Bars show the global proportion of total forestation site area classified as correctly placed (inside) or misplaced (outside) under each framework.*

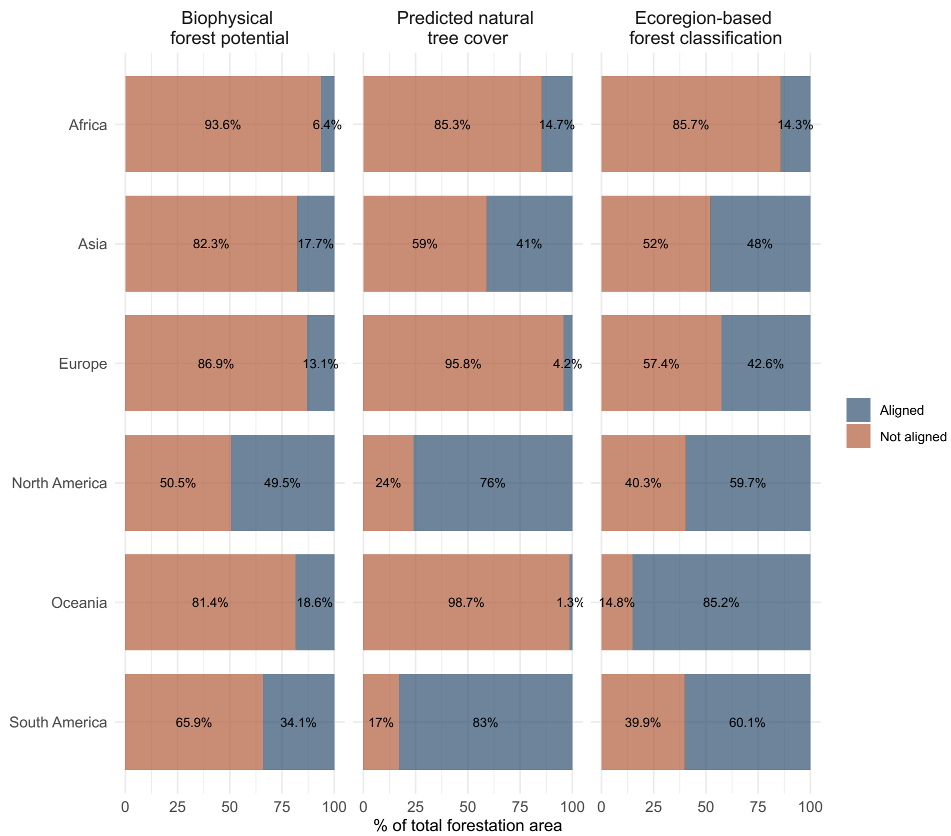

Figure S 3: Continental distribution of the proportion of forestation area classified as correctly placed or misplaced, by framework (biophysical forest potential, predicted natural tree cover, and ecoregion-based forest classification). *Bars show the proportion of total forestation site area within each continent classified as correctly placed (inside) or misplaced (outside) under each framework. Percentages are calculated relative to total forestation area per continent.*

Table S 6: Total forestation area with no overlap with areas identified as forest-dominated ecosystems, by framework.

Continent	Percentage of total area			Area (km^2)		
	Biophysical forest potential	Predicted natural tree cover	Forested ecoregions	Biophysical forest potential	Predicted natural tree cover	Forested ecoregions
Africa	28.88	58.89	52.97	619,907	1,263,799	1,136,910
Asia	40.21	50.19	32.11	260,060	324,619	207,662
Europe	0.98	83.89	0.07	2,340	201,306	169
North America	5.73	8.93	8.80	21,856	34,070	33,600
Oceania	18.12	98.14	14.78	978	5,296	797
South America	6.75	6.65	25.73	47,151	46,491	179,745
Global	**23.12**	**45.54**	**37.85**	**952,293**	**1,875,581**	**1,558,880**

Table S 7: Median predicted natural vegetation cover of forestation area by continent. *Medians and inter-quartile ranges of each class are calculated separately.*

Continent	Median tree (with IQR)	Median short vegetation (with IQR)	Median bare ground (with IQR)
Africa	0.53 (0.36–0.68)	0.45 (0.32–0.59)	0.00 (0.00–0.03)
Asia	0.43 (0.32–0.70)	0.38 (0.28–0.40)	0.12 (0.02–0.29)
Europe	0.53 (0.45–0.58)	0.46 (0.41–0.52)	0.02 (0.01–0.03)
North America	0.84 (0.72–0.93)	0.16 (0.07–0.26)	0.00 (0.00–0.01)
Oceania	0.56 (0.56–0.61)	0.42 (0.38–0.43)	0.01 (0.01–0.01)
South America	0.76 (0.71–0.85)	0.23 (0.14–0.28)	0.01 (0.00–0.01)
Global	**0.54 (0.33–0.74)**	**0.36 (0.25–0.40)**	**0.06 (0.01–0.27)**

Table S 8: Ecoregions overlapping with forestation projects. *For each ecoregion, we report the total forestation area (km^2), number of unique planting sites, continent, and ecoregion category (forest, mosaic, open, treeless).*

Ecoregion	Total forestation area (km^2)	Number of sites	Continent	Category
Dry miombo woodlands	235,446.70	976	Africa	Open
Cerrado	223,642.15	143	South America	Open
Central Zambezian wet miombo woodlands	195,938.10	433	Africa	Open
Zambezian Baikiaea woodlands	177,534.27	307	Africa	Open
Zambezian flooded grasslands	177,069.56	343	Africa	Treeless
Southern Acacia-Commiphora bushlands and thickets	165,944.90	51	Africa	Open
Albertine Rift montane forests	150,926.47	263	Africa	Forest
West Sudanian savanna	120,833.47	224	Africa	Open
Victoria Basin forest-savanna	117,002.10	464	Africa	Mosaic
Guinean forest-savanna	111,062.79	908	Africa	Mosaic
Aravalli west thorn scrub forests	91,344.80	305	Asia	Open
Northern Acacia-Commiphora bushlands and thickets	87,074.88	61	Africa	Open
Madagascar subhumid forests	75,897.77	75	Africa	Forest
Northern Congolian Forest-Savanna	67,331.00	44	Africa	Mosaic
Madeira-Tapajós moist forests	66,508.73	58	South America	Forest
Baltic mixed forests	65,252.83	18	Europe	Forest
Tocantins/Pindare moist forests	57,205.92	55	South America	Forest

Ecoregion	Total forestation area (km^2)	Number of sites	Continent	Category
Mato Grosso tropical dry forests	55,682.30	36	South America	Forest
Scandinavian Montane Birch forest and grasslands	50,275.68	1	Europe	Treeless
New England-Acadian forests	47,883.86	108	North America	Forest
Bahia coastal forests	47,781.84	6	South America	Forest
Madagascar humid forests	46,928.11	164	Africa	Forest
Maranhão Babaçu forests	44,603.79	29	South America	Forest
East Sudanian savanna	42,058.93	1,058	Africa	Open
Central European mixed forests	40,944.23	201	Europe	Forest
Eastern Great Lakes lowland forests	39,913.31	23	North America	Forest
Northern Swahili coastal forests	38,442.86	149	Africa	Forest
Sundarbans mangroves	37,861.45	64	Asia	Forest
Zambezian-Limpopo mixed woodlands	37,065.78	22	Africa	Open
Qin Ling Mountains deciduous forests	36,626.15	3,854	Asia	Forest
Mid-Canada Boreal Plains forests	34,975.57	15	North America	Forest
Central American Atlantic moist forests	34,875.24	1,841	North America	Forest
European Atlantic mixed forests	33,566.59	536	Europe	Forest
Somali Acacia-Commiphora bushlands and thickets	33,338.54	709	Africa	Open
Lower Gangetic Plains moist deciduous forests	33,173.84	74	Asia	Forest
Alto Paraná Atlantic forests	31,179.94	29	South America	Forest
Zambezian mopane woodlands	31,176.80	24	Africa	Open
Northeast US Coastal forests	26,952.54	39	North America	Forest
Deccan thorn scrub forests	24,086.87	5	Asia	Open
Baluchistan xeric woodlands	23,059.78	9	Asia	Open
Southern Rift Montane forest-grassland	22,281.16	9	Africa	Mosaic
Alashan Plateau semi-desert	20,806.00	8,678	Asia	Treeless
Scandinavian and Russian taiga	20,701.33	11	Europe	Forest
Arabian desert	20,101.87	16	Asia	Treeless
Valdivian temperate forests	20,075.26	8	South America	Forest
Mid-Atlantic US coastal savannas	19,721.07	8	North America	Open
Indus Valley desert	19,479.40	8	Asia	Treeless
Chiquitano dry forests	19,021.09	18	South America	Forest
Meghalaya subtropical forests	16,313.27	4	Asia	Forest
Sundarbans freshwater swamp forests	16,215.31	12	Asia	Forest
Tian Shan foothill arid steppe	15,583.19	2	Asia	Treeless
Central American dry forests	15,494.24	113	North America	Forest
Southwest Amazon moist forests	15,036.34	61	South America	Forest
Southern Great Lakes forests	14,741.50	39	North America	Forest
Sahelian Acacia savanna	14,650.03	779	Africa	Open
Madagascar dry deciduous forests	14,612.93	164	Africa	Forest
Changjiang Plain evergreen forests	14,552.43	58,414	Asia	Forest
East African montane forests	14,250.19	58	Africa	Forest
Ethiopian montane grasslands and woodlands	14,230.13	4,343	Africa	Mosaic
Qilian Mountains subalpine meadows	13,868.04	3	Asia	Treeless

Ecoregion	Total forestation area (km²)	Number of sites	Continent	Category
Central Deccan Plateau dry deciduous forests	13,819.27	3	Asia	Forest
Sundaland heath forests	12,681.80	43	Asia	Forest
East Deccan moist deciduous forests	12,549.62	2	Asia	Forest
Southeast US conifer savannas	11,221.68	1,465	North America	Open
Isthmian-Pacific moist forests	11,043.65	659	North America	Forest
Uatumã-Trombetas moist forests	10,876.20	4	South America	Forest
Appalachian-Blue Ridge forests	10,865.38	33	North America	Forest
Hispaniolan moist forests	10,262.17	40	North America	Forest
Guizhou Plateau broadleaf and mixed forests	10,226.93	9,899	Asia	Forest
Eastern Mediterranean conifer-broadleaf forests	10,148.23	28	Asia	Forest
Angolan mopane woodlands	10,096.72	5	Africa	Open
Southern Congolian forest-savanna	9,728.26	10	Africa	Mosaic
Gissaro-Alai open woodlands	9,620.76	1	Asia	Open
Amazon-Orinoco-Southern Caribbean mangroves	9,321.95	66	South America	Forest
Humid Chaco	8,895.28	723	South America	Mosaic
Petén-Veracruz moist forests	8,818.28	1,407	North America	Forest
Guinean mangroves	8,781.10	10	Africa	Forest
Humid Pampas	8,711.79	19	South America	Treeless
Tian Shan montane steppe and meadows	8,674.34	2	Asia	Treeless
Sumatran lowland rain forests	8,308.73	908	Asia	Forest
Sumatran montane rain forests	8,115.15	162	Asia	Forest
Ethiopian montane forests	7,932.07	65	Africa	Forest
Central Tibetan Plateau alpine steppe	7,556.61	3	Asia	Treeless
Cardamom Mountains rain forests	7,464.56	5,493	Asia	Forest
Magdalena-Urabá moist forests	7,352.49	50	South America	Forest
Central African mangroves	7,260.55	18	Africa	Forest
Ucayali moist forests	7,036.43	17,906	South America	Forest
Zambezian evergreen dry forests	6,934.07	26	Africa	Forest
Yunnan Plateau subtropical evergreen forests	6,893.48	10,021	Asia	Forest
Eastern Guinean forests	6,441.36	1,675	Africa	Forest
Costa Rican seasonal moist forests	6,292.18	2,646	North America	Forest
Southwest Borneo freshwater swamp forests	6,276.26	172	Asia	Forest
Luang Prabang montane rain forests	6,064.65	848	Asia	Forest
Trans-Mexican Volcanic Belt pine-oak forests	6,039.05	233	North America	Forest
Northern Indochina subtropical forests	5,837.67	13	Asia	Forest
Borneo lowland rain forests	5,787.27	290	Asia	Forest
Qaidam Basin semi-desert	5,763.51	2	Asia	Treeless
Iberian conifer forests	5,694.98	42	Europe	Forest
Jos Plateau forest-grassland	5,583.93	6	Africa	Mosaic
Albany thickets	5,461.35	1	Africa	Treeless
Central China Loess Plateau mixed forests	5,370.59	50,626	Asia	Forest
Western European broadleaf forests	5,322.81	108	Europe	Forest
Western Congolian forest-savanna	5,206.51	20	Africa	Mosaic

Ecoregion	Total forestation area (km²)	Number of sites	Continent	Category
East Deccan dry-evergreen forests	4,987.47	4	Asia	Forest
Central American pine-oak forests	4,946.87	3,149	North America	Forest
Peruvian Yungas	4,910.73	1,135	South America	Forest
Godavari-Krishna mangroves	4,878.82	12	Asia	Forest
Northwest Congolian lowland forests	4,871.28	5	Africa	Forest
Southern Swahili coastal forests and woodlands	4,806.63	43	Africa	Mosaic
Khathiar-Gir dry deciduous forests	4,618.74	2	Asia	Forest
Nyanga-Chimanimani Montane forest-grassland	4,336.04	4	Africa	Mosaic
Myanmar coastal rain forests	4,256.64	560	Asia	Forest
Eastern Canadian Forest-Boreal transition	4,031.71	17	North America	Forest
Malabar Coast moist forests	3,992.98	3	Asia	Forest
Red Sea-Arabian Desert shrublands	3,942.35	11	Asia	Treeless
Central Andean puna	3,895.39	39	South America	Treeless
Irrawaddy freshwater swamp forests	3,815.37	1	Asia	Forest
Southeast Tibet shrublands and meadows	3,761.17	105,040	Asia	Treeless
Rodope montane mixed forests	3,696.98	2	Europe	Forest
Southern Andean Yungas	3,655.57	20	South America	Forest
Isthmian-Atlantic moist forests	3,541.45	798	North America	Forest
Appalachian Piedmont forests	3,504.52	241	North America	Forest
Western Java rain forests	3,494.41	54	Asia	Forest
North Western Ghats moist deciduous forests	3,416.03	2	Asia	Forest
Eastern Australian temperate forests	3,403.90	3,838	Oceania	Forest
Xingu-Tocantins-Araguaia moist forests	3,377.47	11	South America	Forest
Marajó várzea	3,320.42	3	South America	Forest
Northwest Andean montane forests	3,316.41	15,373	South America	Forest
Mediterranean dry woodlands and steppe	3,297.25	4	Africa	Open
Cameroon Highlands forests	3,205.00	14	Africa	Forest
Talamancan montane forests	3,193.34	286	North America	Forest
East African halophytics	3,167.25	4	Africa	Treeless
Iberian sclerophyllous and semi-deciduous forests	3,068.99	610	Europe	Forest
East African montane moorlands	3,063.26	3	Africa	Treeless
Bahia interior forests	3,009.27	5	South America	Forest
Ethiopian montane moorlands	2,892.34	129	Africa	Treeless
Gulf of California xeric scrub	2,887.63	3	North America	Treeless
Cauca Valley montane forests	2,790.75	1,756	South America	Forest
Central American montane forests	2,774.13	55	North America	Forest
Borneo peat swamp forests	2,764.74	35	Asia	Forest
Zambezian coastal flooded savanna	2,740.16	1	Africa	Mosaic
Mesoamerican Gulf-Caribbean mangroves	2,716.98	23	North America	Forest
Western Java montane rain forests	2,658.98	305	Asia	Forest
Central Indochina dry forests	2,634.89	89	Asia	Forest
Central Andean wet puna	2,630.76	32	South America	Treeless
Monte Alegre várzea	2,499.89	3	South America	Forest

Ecoregion	Total forestation area (km^2)	Number of sites	Continent	Category
Panamanian dry forests	2,454.79	2	North America	Forest
Hispaniolan pine forests	2,436.99	16	North America	Forest
North Arabian desert	2,434.03	9	Asia	Treeless
Arabian sand desert	2,356.34	17	Asia	Treeless
Southern Mesoamerican Pacific mangroves	2,337.12	14	North America	Forest
Eastern Cordillera Real montane forests	2,327.88	213	South America	Forest
Central Afghan Mountains xeric woodlands	2,295.39	2	Asia	Open
Klamath-Siskiyou forests	2,261.59	63	North America	Forest
Hispaniolan dry forests	2,233.96	19	North America	Forest
Myanmar Coast mangroves	2,226.54	290	Asia	Forest
Sri Lanka dry-zone dry evergreen forests	2,189.98	2,270	Asia	Forest
Caledon conifer forests	2,172.67	26	Europe	Forest
Serra do Mar coastal forests	2,134.14	61	South America	Forest
Eastern Arc forests	2,061.59	142	Africa	Forest
Jian Nan subtropical evergreen forests	1,973.63	2,974	Asia	Forest
Kayah-Karen montane rain forests	1,895.21	100	Asia	Forest
Cross-Sanaga-Bioko coastal forests	1,818.65	13	Africa	Forest
Magdalena Valley montane forests	1,772.11	180	South America	Forest
Western Guinean lowland forests	1,739.98	1,319	Africa	Forest
Tumbes-Piura dry forests	1,729.44	18	South America	Forest
California montane chaparral and woodlands	1,710.47	4	North America	Mosaic
Llanos	1,679.59	81	South America	Open
Sierra Madre del Sur pine-oak forests	1,673.56	3	North America	Forest
Borneo montane rain forests	1,638.93	93	Asia	Forest
Qilian Mountains conifer forests	1,601.05	8,681	Asia	Forest
Southern Atlantic Brazilian mangroves	1,591.76	25	South America	Forest
Napo moist forests	1,568.31	40	South America	Forest
Rwenzori-Virunga montane moorlands	1,549.42	5	Africa	Treeless
Balkan mixed forests	1,541.78	14	Europe	Forest
Yucatán moist forests	1,502.96	148	North America	Forest
Orissa semi-evergreen forests	1,502.37	3	Asia	Forest
Himalayan subtropical broadleaf forests	1,480.27	18	Asia	Forest
East African mangroves	1,452.61	11	Africa	Forest
Central Canadian Shield forests	1,451.25	4	North America	Forest
Colorado Rockies forests	1,419.82	149	North America	Forest
Tapajós-Xingu moist forests	1,418.42	1	South America	Forest
Mindanao-Eastern Visayas rain forests	1,401.17	32	Asia	Forest
North Western Ghats montane rain forests	1,394.56	2	Asia	Forest
Willamette Valley oak savanna	1,353.72	8	North America	Open
Dry Chaco	1,313.98	132	South America	Open
Western Himalayan subalpine conifer forests	1,288.87	6	Asia	Forest
Iquitos várzea	1,277.13	43	South America	Forest
Rann of Kutch seasonal salt marsh	1,211.99	424	Asia	Treeless

Ecoregion	Total forestation area (km²)	Number of sites	Continent	Category
Sierra Madre de Chiapas moist forests	1,210.91	121	North America	Forest
Amur meadow steppe	1,169.56	979	Asia	Treeless
South American Pacific mangroves	1,162.98	1	South America	Forest
Cross-Niger transition forests	1,130.33	2	Africa	Forest
Qionglai-Minshan conifer forests	1,121.06	9,485	Asia	Forest
Chiapas montane forests	1,119.97	2	North America	Forest
Madagascar succulent woodlands	1,102.29	58	Africa	Open
Southwest Arabian Escarpment shrublands and woodlands	1,083.74	5	Asia	Mosaic
Northern Rockies conifer forests	1,064.73	14	North America	Forest
Northeast Brazil restingas	1,059.82	4	South America	Mosaic
Terai-Duar savanna and grasslands	1,044.96	6	Asia	Mosaic
California coastal sage and chaparral	1,011.36	14	North America	Mosaic
Tasmanian temperate rain forests	1,005.12	12	Oceania	Forest
Huang He Plain mixed forests	975.08	167	Asia	Forest
Djibouti xeric shrublands	961.55	477	Africa	Treeless
Cuban moist forests	924.62	2	North America	Forest
Sichuan Basin evergreen broadleaf forests	919.64	27,456	Asia	Forest
Scandinavian coastal conifer forests	912.01	1	Europe	Forest
Ordos Plateau steppe	869.39	48,225	Asia	Treeless
Madagascar ericoid thickets	824.43	3	Africa	Treeless
Pantanos de Centla	812.54	132	North America	Mosaic
Luzon rain forests	800.47	24	Asia	Forest
Cauca Valley dry forests	788.88	88	South America	Forest
Sumatran peat swamp forests	774.08	8	Asia	Forest
English Lowlands beech forests	771.33	9	Europe	Forest
Atlantic Coast restingas	756.85	3	South America	Mosaic
North Saharan Xeric Steppe and Woodland	755.49	3	Africa	Open
Southern Pacific dry forests	754.54	5	North America	Forest
Indus River Delta-Arabian Sea mangroves	752.60	231	Asia	Forest
South China-Vietnam subtropical evergreen forests	733.78	748	Asia	Forest
Madagascar mangroves	723.43	5	Africa	Forest
Niger Delta swamp forests	717.34	4	Africa	Forest
Bahamian-Antillean mangroves	696.29	9	North America	Forest
Purus-Madeira moist forests	694.38	1	South America	Forest
Northern Andean páramo	673.56	190	South America	Treeless
Mongolian-Manchurian grassland	671.24	6,152	Asia	Treeless
Suiphun-Khanka meadows and forest meadows	655.83	1,275	Asia	Mosaic
Guajira-Barranquilla xeric scrub	629.33	17	South America	Treeless
Northern Shortgrass prairie	614.76	3	North America	Treeless
Eastern Australia mulga shrublands	611.93	15	Oceania	Treeless
Ecuadorian dry forests	601.69	164	South America	Forest
Japurá-Solimões-Negro moist forests	591.00	1	South America	Forest

Ecoregion	Total forestation area (km²)	Number of sites	Continent	Category
Western Himalayan broadleaf forests	585.68	2	Asia	Forest
Central-Southern Cascades Forests	584.19	91	North America	Forest
Sulawesi lowland rain forests	576.89	3,417	Asia	Forest
Timor and Wetar deciduous forests	569.76	9	Asia	Forest
Miskito pine forests	566.30	2	North America	Open
Tonle Sap-Mekong peat swamp forests	545.29	10	Asia	Forest
North Atlantic moist mixed forests	541.20	10	Europe	Forest
Uruguayan savanna	540.46	1,115	South America	Open
Caatinga	537.24	17	South America	Mosaic
Northern California coastal forests	535.43	2	North America	Forest
Eastern Himalayan broadleaf forests	506.98	12	Asia	Forest
High Monte	503.85	1	South America	Treeless
Tonle Sap freshwater swamp forests	491.47	2	Asia	Forest
Angolan wet miombo woodlands	480.89	4	Africa	Open
Chiapas Depression dry forests	466.59	233	North America	Forest
Belizian pine savannas	440.53	1	North America	Open
Serengeti volcanic grasslands	435.25	2	Africa	Treeless
Kazakh steppe	423.90	4	Asia	Treeless
Manchurian mixed forests	422.52	9,792	Asia	Forest
Syrian xeric grasslands and shrublands	420.14	8	Asia	Treeless
Kalahari xeric savanna	401.93	5	Africa	Open
Caucasus mixed forests	401.14	15	Asia	Forest
Chilean Matorral	389.39	3	South America	Mosaic
Mindanao montane rain forests	365.86	3	Asia	Forest
Western Ecuador moist forests	349.52	35	South America	Forest
Drakensberg Escarpment savanna and thicket	347.45	2	Africa	Open
San Lucan xeric scrub	333.95	1	North America	Treeless
Northeast Spain and Southern France Mediterranean forests	328.98	235	Europe	Forest
Lesser Sundas deciduous forests	320.80	96	Asia	Forest
Bajío dry forests	310.76	16	North America	Forest
Appalachian mixed mesophytic forests	310.42	66	North America	Forest
Fraser Plateau and Basin conifer forests	305.00	44	North America	Forest
Sunda Shelf mangroves	291.20	2,092	Asia	Forest
Bolivian montane dry forests	291.11	37	South America	Forest
Selenge-Orkhon forest steppe	282.22	3	Asia	Mosaic
Mediterranean Acacia-Argania dry woodlands and succulent thickets	280.18	8	Africa	Open
Kwazulu Natal-Cape coastal forests	253.08	1	Africa	Forest
Ghorat-Hazarajat alpine meadow	237.59	1	Asia	Treeless
Zagros Mountains forest steppe	235.43	5	Asia	Mosaic
Southwest Iberian Mediterranean sclerophyllous and mixed forests	226.81	543	Europe	Forest
Changbai Mountains mixed forests	225.61	2,317	Asia	Forest

Ecoregion	Total forestation area (km²)	Number of sites	Continent	Category
Aegean and Western Turkey sclerophyllous and mixed forests	221.21	10	Europe	Forest
Cantabrian mixed forests	214.85	656	Europe	Forest
Daba Mountains evergreen forests	200.22	7,763	Asia	Forest
Himalayan subtropical pine forests	197.78	12	Asia	Forest
Sinú Valley dry forests	195.12	383	South America	Forest
Blue Mountains forests	182.24	7	North America	Forest
Veracruz moist forests	178.92	26	North America	Forest
Nigerian lowland forests	178.70	5	Africa	Forest
Mulanje Montane forest-grassland	178.30	1	Africa	Mosaic
Northern Thailand-Laos moist deciduous forests	177.00	8	Asia	Forest
Southern Annamites montane rain forests	175.56	69	Asia	Forest
Cuban cactus scrub	171.96	1	North America	Treeless
Sri Lanka montane rain forests	162.01	28	Asia	Forest
Cuban pine forests	161.79	2	North America	Forest
Santa Marta montane forests	159.78	184	South America	Forest
Northeast India-Myanmar pine forests	154.48	1	Asia	Forest
Sri Lanka lowland rain forests	151.81	26	Asia	Forest
Balsas dry forests	148.25	3	North America	Forest
Pontic steppe	147.75	16	Europe	Treeless
Sechura desert	146.12	21	South America	Treeless
Madagascar spiny thickets	143.02	90	Africa	Treeless
Southern New Guinea lowland rain forests	141.04	1	Asia	Forest
Pantanal	131.95	14	South America	Mosaic
Hawai'i tropical dry forests	127.08	4	North America	Forest
Eastern Java-Bali rain forests	126.77	76	Asia	Forest
Southwest Arabian coastal xeric shrublands	126.25	2	Asia	Treeless
Northeast Congolian lowland forests	124.97	41	Africa	Forest
Purus várzea	123.85	2	South America	Forest
Ozark Mountain forests	107.88	1,826	North America	Forest
Chocó–Darién moist forests	107.34	34	North America	Forest
Canadian Aspen forests and parklands	106.05	39	North America	Mosaic
Paraná flooded savanna	103.72	2	South America	Treeless
Mesopotamian shrub desert	102.93	2	Asia	Treeless
Central Range Papuan montane rain forests	101.66	1	Asia	Forest
Northern Tallgrass prairie	99.27	2	North America	Treeless
Southeast Australia temperate forests	98.82	150	Oceania	Forest
Brigalow tropical savanna	97.12	6	Oceania	Open
Thar desert	97.06	1	Asia	Treeless
Chihuahuan desert	96.83	21	North America	Treeless
Western Himalayan alpine shrub and meadows	94.67	1	Asia	Treeless
Azerbaijan shrub desert and steppe	93.41	3	Asia	Treeless
Celtic broadleaf forests	91.75	108	Europe	Forest

Ecoregion	Total forestation area (km^2)	Number of sites	Continent	Category
California interior chaparral and woodlands	90.21	21	North America	Mosaic
Etosha Pan halophytics	90.06	1	Africa	Treeless
Southeast Indochina dry evergreen forests	87.06	64	Asia	Forest
Southwest Arabian montane woodlands and grasslands	83.51	4	Asia	Open
Northern Vietnam lowland rain forests	82.39	3	Asia	Forest
Okanogan dry forests	82.05	1	North America	Mosaic
Fynbos shrubland	80.52	2	Africa	Treeless
Western Great Lakes forests	80.16	30	North America	Forest
Luzon montane rain forests	79.06	1	Asia	Forest
Congolian coastal forests	76.02	1	Africa	Forest
Tenasserim-South Thailand semi-evergreen rain forests	75.10	238	Asia	Forest
Cuban dry forests	74.89	1	North America	Forest
Southern New Guinea freshwater swamp forests	74.38	1	Asia	Forest
Northwest Iberian montane forests	72.41	1,071	Europe	Forest
Meseta Central matorral	72.21	44	North America	Treeless
Sierra de la Laguna dry forests	71.11	1	North America	Forest
Northeast China Plain deciduous forests	70.94	14,296	Asia	Forest
Maputaland coastal forests and woodlands	70.20	2	Africa	Mosaic
Atlantic coastal pine barrens	68.59	7	North America	Mosaic
Southeast Papuan rain forests	62.42	1	Oceania	Forest
Saharan halophytics	61.62	1	Africa	Treeless
Montana Valley and Foothill grasslands	60.22	3	North America	Treeless
Veracruz montane forests	57.88	1	North America	Forest
Guinean montane forests	56.83	17	Africa	Forest
Hawai'i tropical low shrublands	55.65	3	North America	Treeless
Sierra de los Tuxtlas	55.02	4	North America	Forest
Sierra Madre Oriental pine-oak forests	46.03	81	North America	Forest
Eastern Java-Bali montane rain forests	42.53	26	Asia	Forest
Cordillera Central páramo	41.18	41	South America	Treeless
Apure-Villavicencio dry forests	38.06	23	South America	Forest
Mizoram-Manipur-Kachin rain forests	37.81	2	Asia	Forest
South American Pacific mangroves	37.20	4	North America	Forest
Sierra Nevada forests	34.56	79	North America	Forest
Da Hinggan-Dzhagdy Mountains conifer forests	29.90	2,101	Asia	Forest
Low Monte	28.87	2	South America	Treeless
Succulent Karoo xeric shrublands	28.23	1	Africa	Treeless
Mediterranean conifer and mixed forests	28.21	1	Africa	Forest
Southwest Australia savanna	28.01	9	Oceania	Open
Jamaican moist forests	27.25	4	North America	Forest
Alai-Western Tian Shan steppe	26.98	2	Asia	Treeless
Cordillera La Costa montane forests	26.90	5	South America	Forest

Ecoregion	Total forestation area (km²)	Number of sites	Continent	Category
Namibian savanna woodlands	26.61	2	Africa	Open
North Arabian highland shrublands	26.42	1	Asia	Treeless
West Sahara desert	26.24	1	Africa	Treeless
Chocó–Darién moist forests	25.36	2	South America	Forest
Eyre and York mallee	24.89	12	Oceania	Open
Eastern Cascades forests	24.78	13	North America	Forest
Angolan montane forest-grassland	24.38	2	Africa	Mosaic
Junggar Basin semi-desert	23.03	4	Asia	Treeless
Hawai'i tropical high shrublands	20.73	1	North America	Treeless
Angolan scarp savanna and woodlands	20.24	3	Africa	Open
Dinaric Mountains mixed forests	19.78	2	Europe	Forest
Sonoran desert	19.74	6	North America	Treeless
Northern Annamites rain forests	19.23	133	Asia	Forest
Patagonian steppe	18.89	2	South America	Treeless
Po Basin mixed forests	18.85	45	Europe	Forest
Caqueta moist forests	17.46	3	South America	Forest
Murray-Darling woodlands and mallee	15.89	12	Oceania	Open
Yucatán dry forests	15.18	40	North America	Forest
Southeast Australia temperate savanna	15.01	13	Oceania	Open
Brahmaputra Valley semi-evergreen forests	14.03	2	Asia	Forest
Mediterranean woodlands and forests	13.94	228	Africa	Forest
Iceland boreal birch forests and alpine tundra	13.89	4	Europe	Forest
Edwards Plateau savanna	13.89	4	North America	Open
North Deccan dry deciduous forests	13.27	1	Asia	Forest
La Costa xeric shrublands	12.93	4	South America	Treeless
Magdalena Valley dry forests	12.52	8	South America	Forest
South Central Rockies forests	11.82	55	North America	Forest
Alberta-British Columbia foothills forests	11.00	1	North America	Forest
Fiji tropical dry forests	10.76	3	Oceania	Forest
Upper Midwest US forest-savanna transition	10.73	7	North America	Mosaic
Sierra Madre Occidental pine-oak forests	10.47	8	North America	Forest
Northern Mesoamerican Pacific mangroves	10.43	3	North America	Forest
Altai steppe and semi-desert	10.42	2	Asia	Treeless
Sinaloan dry forests	10.28	1	North America	Forest
Interior Plateau US Hardwood Forests	10.04	18	North America	Forest
Eastern Himalayan subalpine conifer forests	9.77	7	Asia	Forest
Irrawaddy moist deciduous forests	9.69	2	Asia	Forest
Western Gulf coastal grasslands	9.64	34	North America	Treeless
Cordillera Oriental montane forests	9.52	23	South America	Forest
Nile Delta flooded savanna	9.48	1	Africa	Treeless
Hokkaido montane conifer forests	8.99	2	Asia	Forest
Northern New Guinea lowland rain and freshwater swamp forests	8.49	1	Oceania	Forest

Ecoregion	Total forestation area (km^2)	Number of sites	Continent	Category
Hawai'i tropical moist forests	7.82	3	North America	Forest
Central Pacific Northwest coastal forests	7.78	1	North America	Forest
Indochina mangroves	6.66	29	Asia	Forest
Central Asian riparian woodlands	6.31	4	Asia	Mosaic
Pyrenees conifer and mixed forests	5.88	1	Europe	Forest
Espinal	5.58	31	South America	Open
Greater Negros-Panay rain forests	5.37	28	Asia	Forest
Western shortgrass prairie	5.17	4	North America	Treeless
Horn of Africa xeric bushlands	5.11	2	Africa	Treeless
Peninsular Malaysian montane rain forests	5.02	1	Asia	Forest
Northern Khorat Plateau moist deciduous forests	5.01	264	Asia	Forest
Caatinga Enclaves moist forests	4.47	1	South America	Forest
East European forest steppe	4.33	83	Europe	Mosaic
Queensland tropical rain forests	4.31	61	Oceania	Forest
Sarmatic mixed forests	4.26	109	Europe	Forest
Arizona Mountains forests	4.17	8	North America	Forest
Arabian-Persian Gulf coastal plain desert	4.06	4	Asia	Treeless
Central-Southern US mixed grasslands	3.96	1	North America	Treeless
Southwest Australia woodlands	3.94	42	Oceania	Open
Puerto Rican moist forests	3.88	3	North America	Forest
Guianan piedmont moist forests	3.86	1	South America	Forest
Mount Cameroon and Bioko montane forests	3.56	9	Africa	Forest
Altai montane forest and forest steppe	3.42	1	Asia	Mosaic
Chao Phraya freshwater swamp forests	3.23	16	Asia	Forest
Guianan savanna	2.92	2	South America	Open
Gulf of St. Lawrence lowland forests	2.82	34	North America	Forest
Sumatran freshwater swamp forests	2.80	443	Asia	Forest
Aegean and Western Turkey sclerophyllous and mixed forests	2.76	11	Asia	Forest
Peninsular Malaysian rain forests	2.74	13	Asia	Forest
Richmond temperate forests	2.50	1	Oceania	Forest
Bolivian Yungas	2.44	12	South America	Forest
Carpathian montane forests	2.33	28	Europe	Forest
Palawan rain forests	2.29	2	Asia	Forest
Tibetan Plateau alpine shrublands and meadows	2.22	7	Asia	Treeless
Oaxacan montane forests	2.20	61	North America	Forest
Paropamisus xeric woodlands	2.16	3	Asia	Open
Anatolian conifer and deciduous mixed forests	2.08	5	Asia	Forest
Central US forest-grasslands transition	2.03	4	North America	Mosaic
Motagua Valley thornscrub	1.99	12	North America	Treeless
Central British Columbia Mountain forests	1.87	1	North America	Forest
Alps conifer and mixed forests	1.82	17	Europe	Forest
Limpopo lowveld	1.77	2	Africa	Open

Ecoregion	Total forestation area (km²)	Number of sites	Continent	Category
Irrawaddy dry forests	1.70	1	Asia	Forest
Tamaulipan mezquital	1.64	10	North America	Treeless
South Iran Nubo-Sindian desert and semi-desert	1.62	9	Asia	Treeless
Masai xeric grasslands and shrublands	1.52	2	Africa	Treeless
Southern Anatolian montane conifer and deciduous forests	1.44	6	Asia	Forest
Namaqualand-Richtersveld steppe	1.29	1	Africa	Treeless
Pamir alpine desert and tundra	1.24	1	Asia	Treeless
Italian sclerophyllous and semi-deciduous forests	1.10	74	Europe	Forest
South Arabian plains and plateau desert	1.06	2	Asia	Treeless
Central Mexican matorral	1.04	36	North America	Treeless
Mississippi lowland forests	1.02	1	North America	Forest
Pindus Mountains mixed forests	1.02	61	Europe	Forest
Central Andean dry puna	0.87	1	South America	Treeless
Magellanic subpolar forests	0.85	1	South America	Forest
East Central Texas forests	0.84	3	North America	Forest
Tamaulipan matorral	0.80	63	North America	Treeless
Sayan montane conifer forests	0.79	6	Europe	Forest
Catatumbo moist forests	0.76	1	South America	Forest
Sumatran tropical pine forests	0.75	70	Asia	Forest
Tyrrhenian-Adriatic sclerophyllous and mixed forests	0.75	113	Europe	Forest
Veracruz dry forests	0.73	1	North America	Forest
Central Korean deciduous forests	0.72	28	Asia	Forest
Eastern Panamanian montane forests	0.71	2	North America	Forest
Luzon tropical pine forests	0.68	5	Asia	Forest
Patía valley dry forests	0.63	136	South America	Forest
Pannonian mixed forests	0.56	13	Europe	Forest
Esperance mallee	0.54	15	Oceania	Open
California Central Valley grasslands	0.54	7	North America	Treeless
Venezuelan Andes montane forests	0.53	2	South America	Forest
Ozark Highlands mixed forests	0.47	5	North America	Forest
Eastern Anatolian montane steppe	0.42	3	Asia	Treeless
Jalisco dry forests	0.38	12	North America	Forest
Piney Woods	0.37	2	North America	Mosaic
Great Basin shrub steppe	0.36	4	North America	Treeless
Taiheiyo evergreen forests	0.34	2	Asia	Forest
Solomon Islands rain forests	0.34	1	Oceania	Forest
Sulawesi montane rain forests	0.31	51	Asia	Forest
Kazakh forest steppe	0.29	7	Europe	Treeless
Eastern Himalayan alpine shrub and meadows	0.24	2	Asia	Treeless
Eastern Gobi desert steppe	0.23	2	Asia	Treeless
Red Sea mangroves	0.21	2	Asia	Forest

Ecoregion	Total forestation area (km²)	Number of sites	Continent	Category
Atacama desert	0.19	1	South America	Treeless
Southeast Iberian shrubs and woodlands	0.19	1	Europe	Open
Registan-North Pakistan sandy desert	0.19	2	Asia	Treeless
Baja California desert	0.17	1	North America	Treeless
Palouse prairie	0.13	5	North America	Treeless
Kimberly tropical savanna	0.12	1	Oceania	Open
Sonoran-Sinaloan subtropical dry forest	0.11	1	North America	Forest
South Apennine mixed montane forests	0.11	20	Europe	Forest
Texas blackland prairies	0.10	3	North America	Treeless
Trans-Baikal conifer forests	0.10	1	Asia	Forest
Pernambuco coastal forests	0.10	1	South America	Forest
Snake-Columbia shrub steppe	0.09	3	North America	Treeless
Southwest Arabian highland xeric scrub	0.09	1	Asia	Treeless
Tehuacán Valley matorral	0.08	2	North America	Treeless
Puget lowland forests	0.07	7	North America	Forest
Highveld grasslands	0.07	1	Africa	Treeless
Appenine deciduous montane forests	0.06	4	Europe	Forest
Altai alpine meadow and tundra	0.06	2	Europe	Treeless
West Siberian taiga	0.05	4	Europe	Forest
Central Asian southern desert	0.05	1	Asia	Treeless
Chimalapas montane forests	0.05	1	North America	Forest
Solimões-Japurá moist forests	0.04	2	South America	Forest
East Arabian fog shrublands and sand desert	0.04	1	Asia	Treeless
Mojave desert	0.04	2	North America	Treeless
Cross-Timbers savanna-woodland	0.04	1	North America	Mosaic
Illyrian deciduous forests	0.03	26	Europe	Forest
Southern Vietnam lowland dry forests	0.03	2	Asia	Forest
Nelson Coast temperate forests	0.03	1	Oceania	Forest
Western Siberian hemiboreal forests	0.03	3	Europe	Forest
Central Anatolian steppe and woodlands	0.02	2	Asia	Open
Daurian forest steppe	0.02	1	Asia	Treeless
Chao Phraya lowland moist deciduous forests	0.02	2	Asia	Forest
Central Tallgrass prairie	0.02	2	North America	Treeless
Kazakh steppe	0.01	1	Europe	Treeless
Allegheny Highlands forests	0.01	1	North America	Forest
East Sahara Desert	0.01	1	Africa	Treeless
South Sahara desert	0.01	1	Africa	Treeless
Flinders-Lofty montane woodlands	0.01	1	Oceania	Open
Cyprus Mediterranean forests	0.00	2	Asia	Forest
Negro-Branco moist forests	0.00	1	South America	Forest
Colorado Plateau shrublands	0.00	1	North America	Treeless
Simpson desert	0.00	1	Oceania	Treeless
Flint Hills tallgrass prairie	0.00	1	North America	Treeless

Appendix IV

Supplementary materials to Chapter 5

Table S 9: Categorisation of SANLC classes into three land-cover types*. SANLC data are sourced from the South African National Land Cover dataset (2020), which is provided by the South African Department of Forestry, Fisheries and the Environment (Department of Forestry, Fisheries, and the Environment, 2020). Each SANLC class was categorised into one of three categories: native (which captures the ecosystems that would have existed in the area at the time of Johannesburg's founding, mesic grassy systems with wooded ridges and wetlands); novel, which incorporates ecosystem elements that have been added to the area since the city's establishment (e.g. the extensive urban forest); and urban-industrial, which captures those areas that are largely grey infrastructure (e.g. the inner city).*

Land Cover Class	Categorisation	%
Contiguous Low Forest & Thicket (combined classes)	Native	0.38
Dense Forest & Woodland (35 - 75% cc)	Native	3.85
Open Woodland (10 - 35% cc)	Native	0.76
Contiguous & Dense Planted Forest (combined classes)	Novel	1.49
Open & Sparse Planted Forest	Novel	0.24
Temporary Unplanted Forest	Novel	0.7
Low Shrubland (other regions)	Native	0.06
Sparsely Wooded Grassland (5 - 10% cc)	Native	0.01
Natural Grassland	Native	18.08
Natural Rivers	Native	0.03
Natural Pans (flooded @ obsv time)	Native	0.07
Artificial Dams (incl. canals)	Novel	0.25
Artificial Sewage Ponds	Novel	0.04
Artificial Flooded Mine Pits	Novel	0.05
Herbaceous Wetlands (currently mapped)	Native	2.34
Herbaceous Wetlands (previous mapped extent)	Native	1.14
Natural Rock Surfaces	Native	0.15
Dry Pans	Native	0.02
Bare Riverbed Material	Native	0.03
Other Bare	Urban-industrial	1.09
Cultivated Commercial Permanent Orchards	Novel	0.01
Commercial Annuals Pivot Irrigated	Novel	0.5
Commercial Annuals Non-Pivot Irrigated	Novel	0.04
Commercial Annuals Crops Rain-Fed / Dryland / Non-Irrigated	Novel	2.74
Subsistence / Small-Scale Annual Crops	Novel	0.17
Fallow Land & Old Fields (Trees)	Novel	0.26
Fallow Land & Old Fields (Bush)	Novel	0.03
Fallow Land & Old Fields (Grass)	Novel	3.99
Fallow Land & Old Fields (Bare)	Urban-industrial	0.3
Fallow Land & Old Fields (Low Shrub)	Novel	0.04
Residential Formal (Tree)	Novel	5.13
Residential Formal (Bush)	Novel	0.98
Residential Formal (low veg / grass)	Novel	14.17
Residential Formal (Bare)	Urban-industrial	16.06
Residential Informal (Tree)	Novel	0.04
Residential Informal (Bush)	Novel	0.00
Residential Informal (low veg / grass)	Novel	0.77
Residential Informal (Bare)	Urban-industrial	1.03
Village Scattered (bare only)	Urban-industrial	1.06
Village Dense (bare only)	Urban-industrial	0.25
Smallholdings (Tree)	Novel	1.97
Smallholdings (Bush)	Novel	0.12
Smallholdings (low veg / grass)	Novel	4.02
Smallholdings (Bare)	Urban-industrial	1.13

Land Cover Class	Categorisation	%
Urban Recreational Fields (Tree)	Novel	0.76
Urban Recreational Fields (Bush)	Novel	0.02
Urban Recreational Fields (Grass)	Novel	0.93
Urban Recreational Fields (Bare)	Urban-industrial	0.16
Commercial	Urban-industrial	6.34
Industrial	Urban-industrial	3.7
Roads & Rail (Major Linear)	Urban-industrial	0.37
Mines: Surface Infrastructure	Urban-industrial	0.01
Mines: Extraction Sites: Open Cast & Quarries combined	Urban-industrial	0.59
Mines: Waste (Tailings) & Resource Dumps	Urban-industrial	1.23
Land-fills	Urban-industrial	0.07
Fallow Land & Old Fields (wetlands)	Native	0.25

Table S 10: Percentage of native, novel and urban-industrial ecosystem coverage in each of Johannesburg's 135 census wards. *Each of the three land-cover types (native, novel and urban-industrial) was aggregated from multiple land-cover classes in the South African National Land Cover dataset (see **Table S 9**).*

Ward	Native	Novel	Urban-industrial
1	20.26	17.35	62.21
2	20.26	17.48	62.08
3	10.9	15.92	72.99
4	8.59	21.27	69.95
5	46.71	38.66	14.45
6	48.82	27.41	23.59
7	22.92	23.24	53.66
8	52.28	32.77	14.76
9	47.84	35.2	16.78
10	38.33	41.86	19.62
11	13.61	53.86	32.34
12	9.43	25.71	64.66
13	43.06	35.02	21.73
14	23.87	24.32	51.62
15	6.03	8.92	84.85
16	14.09	17.3	68.42
17	30.18	26.48	43.15
18	28.92	21.32	49.57
19	34.38	16.22	49.21
20	9.25	10.64	79.91
21	16.26	10.24	73.3
22	7.29	26.49	66.03
23	39.84	48.62	11.35
24	37.78	17.83	44.2
25	30.29	16.65	52.87
26	17.99	8.89	72.92
27	4.87	7.41	87.53
28	8.94	11.15	79.72
29	30.76	6.95	62.1
30	7.8	6.27	85.74
31	8.01	7.15	84.62
32	26.02	49.8	23.99
33	12.97	13.39	73.45
34	16.58	13.93	69.3
35	11.73	10.77	77.31
36	27.23	12.63	59.95
37	33.35	16.91	49.55
38	30.92	13.21	55.69

Ward	Native	Novel	Urban-industrial
39	21.05	13.37	65.38
40	8.33	11.22	80.26
41	5.22	6.48	88.1
42	10.31	9.42	80.07
43	6.94	7.57	85.3
44	10.1	9.19	80.53
45	7.63	8.47	83.71
46	18.01	24.19	57.61
47	6.9	6.17	86.73
48	7.57	12.6	79.63
49	29.29	14.77	55.75
50	50.72	15.66	33.42
51	18.55	6.95	74.32
52	7.61	8.75	83.44
53	45.86	36.99	16.96
54	14.24	50.04	35.53
55	15.78	32.99	51.04
56	14.66	42.9	42.25
57	26.59	19.37	53.86
58	10.33	26.01	63.47
59	8.29	1.53	89.98
60	15.62	11.56	72.63
61	13.54	4.84	81.42
62	7.25	3	89.55
63	14.91	0.97	83.92
64	21.2	16.58	62.02
65	18.51	8.45	72.85
66	13	47.1	39.72
67	33.35	23.66	42.79
68	29.97	14.2	55.64
69	14.44	34.35	51.02
70	28.87	30.98	39.95
71	12.19	44.87	42.75
72	12.52	66.76	20.53
73	9.05	61.31	29.46
74	15.02	53.94	30.85
75	1.23	3.02	95.55
76	3.04	2.83	93.93
77	5.42	4.68	89.7
78	6.3	3.36	90.14
79	2.75	6.3	90.75
80	6.84	35.37	57.59
81	22.62	48.41	28.78
82	14.74	25.49	59.58
83	14.45	68.03	17.32
84	10.86	43.57	45.38
85	21.43	50.4	27.98
86	17.25	44.44	38.12
87	11.13	64.41	24.27
88	16.2	68.51	15.11
89	14.91	56.7	28.19
90	12.64	64.46	22.7
91	11.58	39.92	48.3
92	15.54	53.45	30.82
93	14.71	48.2	36.89
94	20.77	56.14	22.89

Ward	Native	Novel	Urban-industrial
95	5.14	16.61	78.05
96	32.59	46.94	20.27
97	20.42	50.96	28.43
98	8.85	66.73	24.23
99	7.85	72.69	19.27
100	23.72	12.03	64.06
101	9.79	48.07	41.94
102	14.6	56.18	29.04
103	10.7	60.35	28.77
104	10.82	62.93	26.06
105	7.91	10.56	81.34
106	14.99	56.15	28.66
107	1.8	1.77	96.23
108	1.04	0.34	98.43
109	25.7	42.71	31.4
110	11.56	50.52	37.72
111	10.55	2.99	86.25
112	11.73	49.96	38.12
113	4.84	48.13	46.83
114	16.65	49.16	33.99
115	11.07	56.53	32.21
116	0.81	1.91	97.08
117	19.13	57.97	22.72
118	19.09	41.43	39.28
119	36.75	39.55	23.51
120	42.36	17.58	39.88
121	58.75	22.28	18.79
122	49.41	41.61	8.79
123	9.46	9.09	81.25
124	13.71	11.1	74.99
125	55.81	29	15.01
126	10.98	61.16	27.67
127	48.47	18.1	33.24
128	12.09	11.61	76.11
129	32.7	13.36	53.75
130	12.79	11.22	75.79
131	11.12	10.96	77.73
132	28.88	38.65	32.28
133	2.89	2.06	94.85
134	9.38	54.11	36.32
135	11.14	64.2	24.47

***Table S 11: Correlations between pairs of provisioning ecosystem services**. Numbers represent Spearman's rank correlation coefficients (ρ), with statistical significance indicated by an asterisk (*) for $p < 0.05$.*

	Wood for cooking	Wood for heating	Livestock production	Poultry production	Vegetable production
Wood for cooking	1				
Wood for heating	0.62*	1			
Livestock production	0.11	0.13	1		
Poultry production	0.19*	0.20*	0.86*	1	
Vegetable production	0.29*	0.34*	0.50*	0.72*	1
Fresh water	0.55*	0.36*	-0.03	0.01	0.02

***Table S 12: Diagnostics for spatial dependence in models.** For each response variable, Moran's I and its p-value were calculated. Spatial error models were fitted where there was evidence of spatial autocorrelation ($p < 0.05$). AIC values were* *reported for original and spatial error models. The coefficients and magnitude were consistent across original and spatial error models, leaving substantive conclusions unchanged*

Response variable	Moran's I	P-value	AIC (non-spatial)	AIC (spatial error)	Interpretation
LST	0.43	< 2.2e-16	369.5	297.95	Coefficients same sign & magnitude
BII	0.26	1.413e-07	-272.94	-288.87	Coefficients same sign & magnitude
Poultry production	0.27	2.298e-08	267.99	249.44	Coefficients same sign & magnitude
Livestock production	0.15	0.0004	276.38	271.39	Coefficients same sign & mostly same magnitude
Vegetable production	0.38	9.54e-15	301.98	250.93	Coefficients same sign & magnitude
Cooking fuel	0.005	0.2964	-142.95	N/A	No spatial correlation
Heating fuel	0.21	3.985e-06	437.09	422.65	Coefficients same sign & magnitude
Water provision	-0.02	0.4805	-303.06	N/A	No spatial correlation

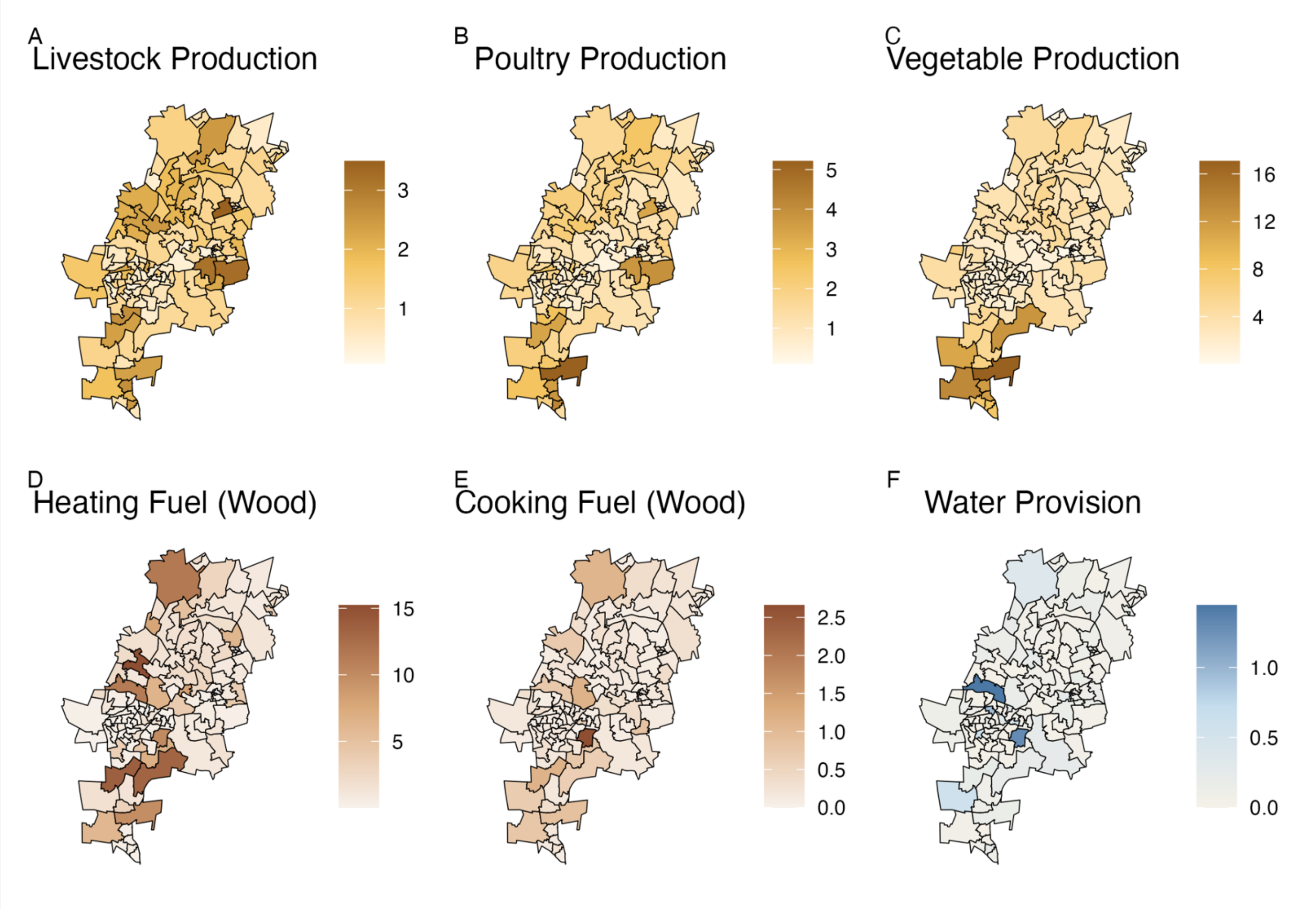

Figure S 4: Spatial distribution of ecosystem services across Johannesburg wards*. The percentage of households in each ward that reported, in the 2011 national census and/or 2016 community survey, making use of the following ecosystem services: A) food production: livestock; B) food production: poultry; C) food production: vegetable; D) fuel provision: wood used as main fuel source for heating; E) fuel provision: wood used as main fuel source for cooking; and F) water provision: main water source being from natural water sources. Panels A–C use a maize-coloured gradient from pale cream to rich golden-brown to indicate increasing food production, panels D–E use a wood-toned gradient from light beige to deep brown to indicate increasing use of wood fuel, and panel F uses a dry-to-wet gradient from light neutral to deeper blue to indicate increasing reliance on natural water sources. The wards on the southern and north-eastern periphery of Johannesburg tend to have higher provisioning ecosystem service use by households.*

*Table S 13: **Full regression model coefficients for the effects of land cover and demographic data on NbS benefits.** Estimates are presented along with standard errors and p-values.*

Response	Predictor	Estimate	Std. Error	P-value
Land surface temperature	Native	-1.82	0.29	4.52E-09
	Novel	-1.44	0.18	1.64E-13
	Income	0.00	0.00	1.37E-08
	White	-1.15	0.70	1.01E-01
	Black African	-0.10	0.67	8.82E-01
Biodiversity intactness	Native	0.27	0.03	2.00E-16
	Novel	0.12	0.02	8.16E-12
	Income	0.00	0.00	3.19E-03
	White	-1.84	1.40	1.92E-01
	Black African	-1.38	1.37	3.19E-01
Provisioning services	Native	2.72	0.45	8.03E-04
	Novel	0.88	0.27	6.29E-02
	Income	0.00	0.00	1.96E-01
	White	-5.18	3.33	1.10E-01
	Black African	-4.33	3.35	1.57E-01
Poultry production †	Native	-0.37	0.11	1.78E-02
	Novel	-0.17	0.09	1.45E-03
	Income	0.00	0.00	1.45E-04
	White	-0.97	0.60	1.65E-01
	Black African	-0.85	0.60	4.01E-01
Livestock production †	Native	-0.36	0.15	7.26E-03
	Novel	-0.33	0.10	8.77E-03
	Income	-1.42E-06	3.62E-07	3.65E-02
	White	1.11	0.80	1.21E-01
	Black African	0.66	0.79	6.90E-03
Vegetable production †	Native	-0.31	0.06	3.91E-01
	Novel	-0.16	0.06	6.68E-01
	Income	1.24E-06	4.75E-07	8.06E-01
	White	1.08	0.52	6.67E-01
	Black African	0.79	0.51	7.64E-05
Cooking fuel †	Native	-3.18	1.16	3.13E-03
	Novel	-1.04	1.20	3.69E-02
	Income	0.00	0.00	4.48E-01
	White	-11.13	45.12	9.34E-01
	Black African	-18.86	43.79	1.51E-02
Heating fuel †	Native	-0.79	0.19	1.94E-01
	Novel	-0.59	0.20	8.99E-02
	Income	0.00	0.00	9.08E-01
	White	0.35	0.46	4.74E-01
	Black African	0.04	0.45	2.00E-16
Water from natural sources	Native	-4.62	1.88	1.37E-12
	Novel	3.51	2.69	7.57E-01
	Income	0.00	0.00	1.33E-08
	White	-1.69	14.65	1.66E-03
	Black African	-10.05	13.99	7.57E-02
Novel cover	Income	1.63E-04	1.09E-08	1.24E-01
	White	0.85	0.12	1.96E-01
	Black African	0.01	0.02	4.52E-09