PBL Netherlands Environmental Assessment Agency

ASSESSING THE IMPACT OF INTERNATIONAL COOPERATIVE INITIATIVES ON BIODIVERSITY

Kok, M. T. J., Schoolenberg, M. A., Löwenhardt, H. M. R., Voora, V., van Oorschot, M., Arlikatti, N., Arts, B.

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Corresponding author
marcel.kok@pbl.nl

Authors
Kok, M. T. J., Schoolenberg, M.A., Löwenhardt, H. M. R., Voora, V., van Oorschot, M., Arlikatti, N., Arts, B.

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Contents

PREFACE

This report by PBL Netherlands Environmental Assessment Agency — in collaboration with the Forest and Nature Conservation Policy group (FNP) of Wageningen University & Research — documents an explorative and methodological analysis of the contribution of voluntary non-state initiatives to the realisation of international biodiversity targets. This research was performed over the 2017–2020 period (and, where possible, updated in 2023) to develop a methodology for assessing the contribution by non-state actors to area-based conservation and for evaluating the interventions and management options they applied. We performed a number of rather diverse case studies with various lifespans in different stages of implementation around the world. This proved to be quite challenging in terms of methodology development and data availability. Furthermore, completion of the report was also affected by the delays of the Kunming-Montreal COP15 negotiations to the Convention on Biological Diversity (CBD) (its initial target audience), the COVID-19 pandemic, and finally also by our own shifting priorities in the 2020–2022 period. We realise that our analysis, at this point, might raise more questions than answers. We nevertheless decided to document our work so far, rather than further strengthening the approach up to a level of 'top' methodological rigour and 'full' case consistency in its application, as this would require further delay and more resources. This document is therefore particularly relevant to be used by colleagues as input material for any subsequent steps in assessing the contribution of voluntary non-state initiatives to the enhancement of biodiversity conservation and its sustainable use.

Summary

The urgent need for biodiversity conservation within the gamut of global environmental issues has gained significant traction over the past three decades. To achieve multilateral targets for biodiversity conservation, as were set out through multiple frameworks established in subsequent Conferences of the Parties (COP) to the Convention on Biological Diversity (CBD), the global community has identified the importance of international environmental governance as an approach. In doing so, the global community has recognised that the onus of undertaking concrete action towards biodiversity conservation cannot solely rely on state actors. Specifically, the successful redressal of biodiversity conservation requires a whole-of-society approach. Consequently, the role of non-state and sub-national actors as change agents towards redressing biodiversity loss has been duly recognised. These actors have demonstrated an ability to play a vital role in filling some of the implementation gaps in global commitments made by state actors.

While non-state and sub-national actors have gained importance in international environmental governance, the focus has shifted from government to governance. In this context, the parties to the Convention on Biological Diversity decided to implement the 'Sharm El Sheikh to Beijing Action Agenda for Nature and People', which was launched in Egypt at CBD's COP14 in November 2018 and continues on to and beyond COP15 in Montreal in 2022. As part of this agenda, countries agreed to invite voluntary biodiversity commitments from non-state and sub-national actors (e.g. cities, regions, civil society groups, religious groups, and companies). In doing so, possibilities for strengthening the role of these actors in the new global biodiversity framework were explored. With the arrival of non-state and sub-national actors for catalytic action and change at the global stage, including over 700 voluntary commitments in the CBD context, came the need for frameworks to assess their contribution, impact and effectiveness. The 'Sharm El Sheikh to Beijing Action Agenda for Nature and People' too has recognised the need for an accountability framework for non-state actors, now included in the monitoring, reporting and review process as part of the Kunming-Montreal Global Biodiversity Framework. This report aims to present a methodology for analysing the contribution, impact and effectiveness of these actors towards the realisation of some of the strategic goals of the CBD, which will be conducted in a twofold manner of enquiry: 1) Analysing the interventions of so-called International Cooperative Initiatives on Biodiversity (ICIBs) that aim at conservation and sustainable use; and 2) Creating a methodological tool — through a joint learning process — for the assessment of biodiversity impacts of non-state actors and voluntary commitments on the ground. This endeavour is far from finished, and the current assessment framework is just a stage in that learning process. We recommend that more research and suggestions are needed on how to assess indirect effects, which we did not do in this study, to avoid a focus on quantitative data and technocratic monitoring only, as well as to avoid overlooking those indirect effects that are also needed for transformative change.

Finally, this report recognises the large diversity of non-state and sub-national actors and our case studies range from integrated landscape management to rewilding initiatives, and from agricultural certification to community forestry. Cases also vary in terms of data availability and certainty, and thus some could be investigated 'deeper' than others. Regardless, we can conclude that these ICIBs already produce substantial outcomes in the number of initiated projects (several thousands) and the amount of land area protected or managed by ICIBs (several 100s of millions of hectares) worldwide. In doing so, this study provides useful insights into how ICIBs may contribute towards positive biodiversity impacts on the ground, and how to assess those.

1 Introduction

In the context of the preparations of the post-2020 Global Biodiversity Framework, decided at COP15 in Montreal, Parties to the Convention on Biological Diversity (CBD) decided to implement the 'Sharm El Sheikh to Beijing Action Agenda for Nature and People', which was launched in Egypt at the CBD 14th Conference of the Parties (COP14) in November 2018. Countries agreed to invite voluntary biodiversity commitments from non-state and sub-national actors (e.g. cities, regions, civil society groups, religious groups and companies), as well as to explore possibilities for strengthening the role of non-state and sub-national actors in the new global biodiversity framework. By late 2022, according to the CBD Action Agenda, the organisation has collected over 700 commitments by almost 300 international partnerships. In the meantime several international networks like Business4Nature, Cities with Nature and IUCN through the Contributions for Nature Platform, have set up portals to collect the commitments from their constituencies.

This approach coincides with a broader development in global environmental governance, in which non-state actors and voluntary commitments by non-state actors are becoming part of the international agreements and United Nations (UN) conventions. Examples of this are recent developments in climate change (Kuyper et al., 2017), Sustainable Development Goals (Kanie and Biermann, 2017) and ocean governance (Neumann and Unger, 2019).

The growing attention for non-state and sub-national actors in international environmental governance in the UN reflects the global shift from government to governance. The absence of sufficient action by states to address global environmental issues, and the combination of globalisation, decentralisation and democratisation (Pierre and Peters, 2000; Pierre, 2000), give way to a wide array of voluntary initiatives, such as voluntary sustainability standards, restoration initiatives, and greening cities projects, that have begun to fill the implementation gap (e.g. Pattberg et al., 2017, 2019; Bäckstrand et al., 2017; UNEP, 2018; Lemos and Agrawal, 2006).

Non-state actors[1] increasingly take up the role of new agents of change through participating in and setting rules related to earth system governance (Biermann et al., 2010). The 'Sharm El Sheikh to Kunming Action Agenda for Nature and People'[2] has the explicit aim to catalyse actions from all sectors and stakeholders in support of biodiversity conservation and its sustainable use (CBD, 2018; see also Kok et al., 2019). An online platform is to be set up to map current global efforts of non-state actors towards biodiversity conservation, in order to assess impact and gaps therein. In addition to the Portal of the CBD Action Agenda and portals by other constituencies, other platforms have emerged that collect area-based commitments for nature, such as the WCMC Nature Commitments Platform. All in all, since the inception of this study we see a proliferation of portals and platforms for non-state commitments for nature.

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[1] For brevity, this report uses the term non-state action/actors as mutually inclusive of sub-national action/actors.

[2] Due to change in location of the COP-15 twice over, from Beijing to Kunming and later to Montreal, on account of the COVID-19 pandemic, the name of the Action Agenda was changed. Hereon we refer to it as CBD Action Agenda.

To ensure the credibility and transparency of the Action Agenda, it is imperative to have clear insights into the role and effectiveness of non-state actors in biodiversity conservation and its sustainability along with methodologies required to assess their impact, i.e. in this context, it is crucial to gauge what International Cooperative Initiatives on Biodiversity (ICIBs) actually do and can contribute to biodiversity gains on the ground. However, no clear methodologies to conduct such analyses currently exist. Therefore, while this report tries to assess the contribution of non-state actions towards biodiversity conservation and its sustainable use, it also explores methods and tools to assess their impacts. This endeavour has become all the more relevant now that the CBD Action Agenda has been integrated in the Kunming-Montreal Global Biodiversity Framework, and non-state actions are now becoming subject to an accountability framework that includes monitoring, reporting, and verification.

Given the critical need to assess ICIB impacts towards biodiversity conservation and sustainable use, while conceiving relevant assessment methodologies, this report is composed around a research objective that is twofold. Firstly, this study aims at quantifying — as far as possible — the contribution of non-state initiatives towards the conservation and sustainable use of biodiversity on the ground. Secondly, it aims at developing a methodology for analysing these contributions and assessing its impact. This methodology will especially focus on analysing the strategic goals of the CBD that aim to strengthen the status and sustainable use of biodiversity, by safeguarding ecosystems, species and genetic diversity, and by improving management interventions. The study will therefore answer the two following research questions:

> *What is the contribution of international cooperative initiatives towards the conservation and sustainable use of biodiversity?*
>
> *What methodology can be applied for the assessment of biodiversity impacts of non-state actors and voluntary commitments, and what challenges may arise from this exercise?*

While recognising that the range of non-state actors is diverse, here the focus is on the biodiversity impacts of large-scale International Cooperative Initiatives on Biodiversity (ICIBs), either in the form of formal organisations or less formal 'movements', which create positive, area based, on-the-ground biodiversity impacts. We have chosen to do so because of the expected larger impact and the feasibility to quantify the impacts of these initiatives. We define ICIBs as initiatives of international, non-state and sub-national actors that are either entirely private (from business or civil society) or have some state involvement (hybrid) (see also Pattberg et al., 2017; 2019). However, because we also recognise non-state international initiatives that lack a clear coordination unit, the analysis was broadened to include initiatives that could be defined as a movement with global relevance for biodiversity conservation. For the sake of simplicity, we will use the term non-state initiatives in the rest of this report to refer to ICIBs, including both such organisations and movements.

Specifically, this report aims to quantify impacts of ICIBs in terms of 'biodiversity gains' or 'avoided loss'. In order to reduce complexity and bring about a more cordial comparability of cases, we decided to do this with an existent single biodiversity indicator — the so-called MSA-indicator (to be explained below) — that is readily applicable globally and sensitive to environmental change, hence suited to capture biodiversity gains. It is obvious, however, that the contributions of ICIBs to biodiversity governance are manifold, and include governance functions such as developing standards and commitments, providing finance, exchanging information and creating networks as well as operational activities that cannot directly be translated into biodiversity impacts on the ground (Pattberg et al., 2017). Furthermore, as it is difficult to capture all qualities of biodiversity in one indicator, we realise the limitations of this choice. The advantage is, however, that the consistent use of a single indicator for quite different cases will facilitate comparability between those.

The analysis in this report focuses on existing ICIBs in society, for which we expected that we would be able to gather sufficient data from a variety of secondary sources to carry out at least a preliminary ex-post evaluation of their contribution to the conservation and sustainable use of biodiversity. However, in the context of the CBD Post-2020 framework, an ex-ante evaluation of voluntary commitments and pledges will be needed to quantify what these commitments might contribute to the goals and targets of the CBD for the post-2020 process. Such an evaluation will furthermore need to analyse if the commitments add up to meet the goals and targets or conversely determine the ambition gap.

This report is organised as follows: Chapter 2 introduces the methodology applied in this report followed by the individual case studies presented from Chapters 3 to 10. Table 2.1 provides an overview of the cases that were selected (see Chapter 2). Chapter 11 presents a discussion and synthesis of the analysis.

2 Methods

2.1 Introduction

The methodology in this report is based on an analysis of data from ICIBs and secondary data from the scientific literature, and consists of four building blocks that will be presented in subsequent sections. First, we define the concept of impact and introduce various methods for impact analysis in which we position the method applied in this report (Section 2.2). Secondly, we explain our method to capture biodiversity impacts in terms of 'Mean Species Abundance' (MSA) combined with systematic literature reviews of available impact studies for the specific ICIB (Section 2.3). Thirdly, we present a framework for assessing non-state actor contributions to on-the-ground biodiversity levels, that we adapted from an earlier methodological exploration (Arts et al., 2017) (Section 2.4). Fourthly, we outline how we have identified the ICIBs that we have analysed as case studies in this report (Section 2.5).

2.2 Defining impacts

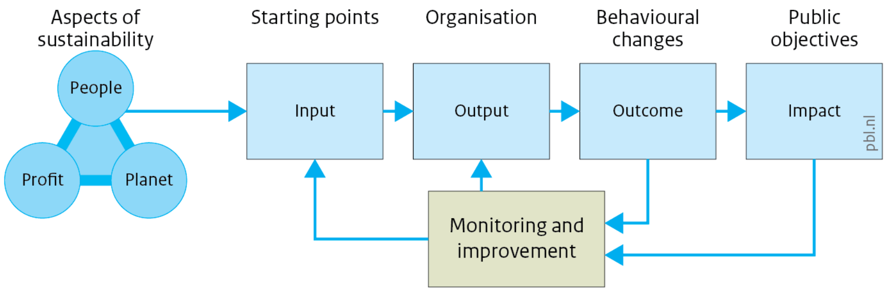
Before describing how we evaluate the impacts on biodiversity as a consequence of actions by ICIBs, we first address how we define 'impact' in itself. A commonly accepted framework to analyse impact in the political sciences literature is to relate it to input, output, and outcome. Such thinking goes back to Easton's political system approach (Easton, 1957), but the input–impact model is widely used in many disciplines, such as organisation studies, conflict studies and international relations (Spar and Dail, 2002; Underdal and Young, 2004; Wolf, 2010). This model can be applied in developing and assessing theories of change and is also often used in ex-ante and ex-post evaluations of environmental governance.

Based on Easton's (1957) understanding of the input–output political system approach, Inputs can be defined as ideas, demands, support and capacity for collective problem solving or opportunity identification. These inputs can either directly relate to intentions to contribute to conserving biodiversity or minimising negative impacts, or this can be a side effect of these inputs. Outputs relate to commitments — expressed in projects, programs, policies, law, funds, etc. — to address these problems and opportunities. Examples of this in biodiversity governance are the creation of standards for agro-commodities, strategic plans aimed at greening cities, or making agreements with landowners. Outcomes can be defined as behavioural changes — i.e. applying management practices that lead to biodiversity gains, based on such commitments. Impacts are the actual, real-life contributions to relieving pressures on biodiversity and realising biodiversity gains, resulting from those behavioural changes.

Next to direct impacts in terms of biodiversity, it is also important to realise that indirect or unforeseen impacts may occur, such as protesting or pressurising governments that undertake actions as a consequence. Or ICIBs may fulfil other governance functions, such as providing information or setting standards, that may have unforeseen positive side effects on the ground. Figure 2.1 shows an image of the I-O-O-I scheme, or the 'implementation chain', taken from a study on sustainable supply chains (van Oorschot et al., 2014, p. 31). It also shows feedback mechanisms among I-O-O-I, namely monitoring and improvement, as already acknowledged in the early literature (Easton, 1957).

Figure 2.1

Assessment framework for the sustainable development of supply chains



Source: Van Tulder, 2010; adaptation by PBL

Implementation chain of the assessment framework

Various approaches exist for performing an impact assessment. Hulme (2000) distinguishes three traditions: (1) the scientific method, based on various quantitative techniques, particularly large-N quasi-experiments; (2) the humanities tradition, based on qualitative methods or mixed methods; and (3) the Participatory Learning and Action (PLA) approach, based on ethnographies and/or action research. While the first tradition tries to **attribute** as accurately as possible an impact to a specific intervention, the third tradition aims at having an **impact itself**, by improving practices through social learning processes, together with stakeholders. The second tradition sits in-between, trying to qualitatively and/or quantitatively assess the **contribution** of a certain intervention to a certain result, as part of a broader set of possible explanatory conditions (Ton et al., 2014).

Given the research objective of this study, to quantify ICIB's contributions and impacts, it would be expected to employ the first tradition based on Hulme (2000). However, quasi-experimental research in this field is scarce, and is yet to be conclusively quantified, if it ever were possible to begin with. Therefore, this study will build upon a combination of both quantitative and qualitative research methods. On the one hand, we support the core idea of mixed methods and triangulation of methods and data, because these can bring both generality and depth to impact analysis at the same time. On the other hand, we simply have to build upon mixed methods, because this study conducts secondary analyses of data and studies from both traditions (i.e. the scientific ones and those from the humanities).

2.3 Biodiversity impacts

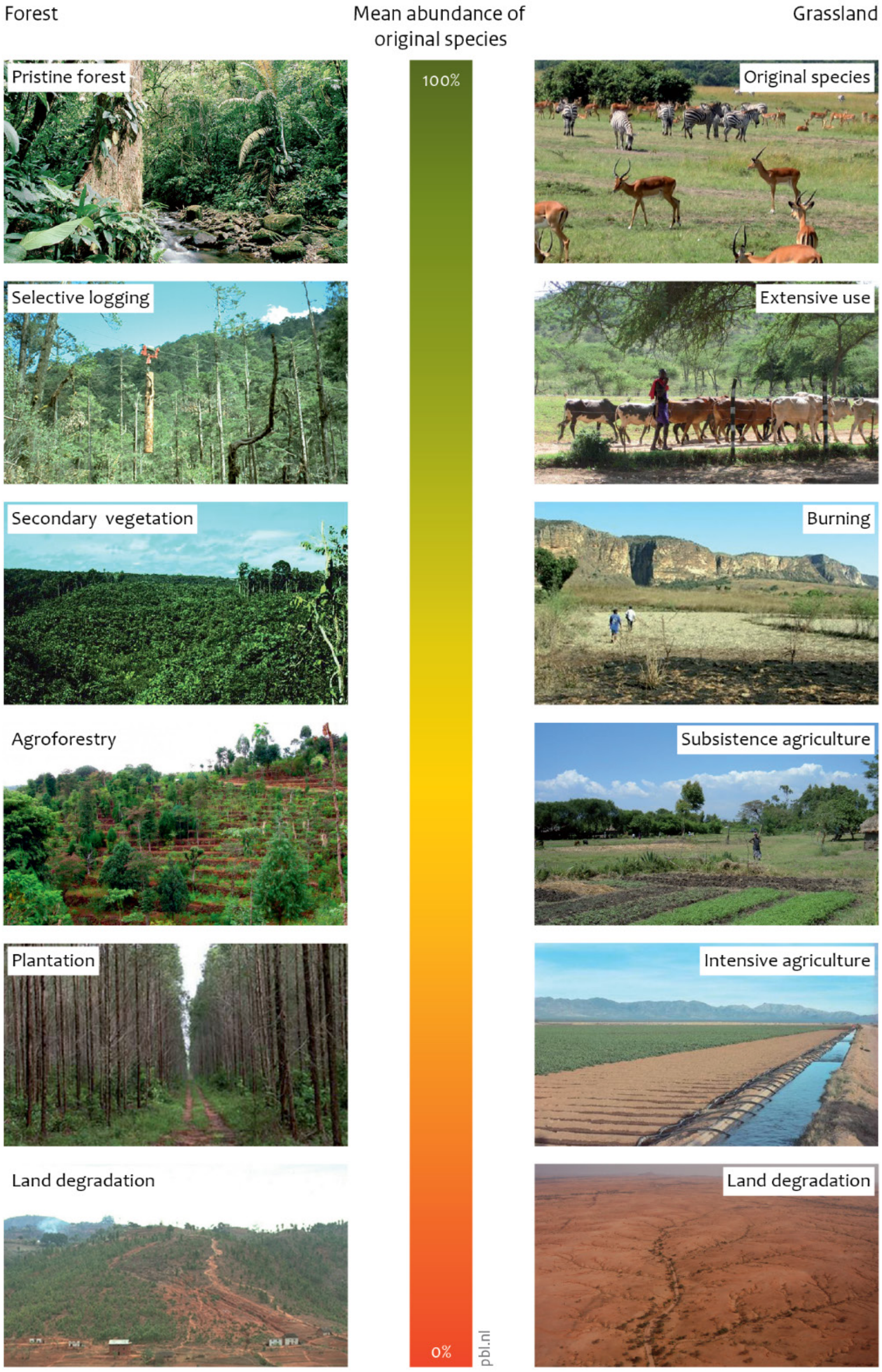
To be able to compare ICIBs in a consistent manner, we evaluate their impacts on the conservation of biodiversity using a single indicator. We use the model relationships within the GLOBIO model (v3.6, see Schipper et al., 2016) to assign Mean Species Abundance (MSA) values to the initiatives to quantify the biodiversity impact potential on the area covered and positively influenced by individual ICIBs.

MSA is based on the abundance of species in an altered ecosystem compared to their abundance in a natural reference situation. MSA values are derived through meta-analysis from empirical studies that compare species assemblages between the pristine state of an ecosystem to a managed or altered state. In this regard, MSA values can be considered an indicator of the 'naturalness' of a modified ecosystem, where a value of 1 means that the species assemblage is fully intact and a value of 0 indicates that all original species are extirpated. The activities of an ICIB (i.e. changes in management practices to achieve improvements in biodiversity) are linked to MSA values related to management categories of a specific land use system, taken from the GLOBIO 3.6 model (Alkemade et al., 2013; Schipper et al., 2016; www.globio.info). For example, in GLOBIO 3.6 (Schipper et al., 2016), a natural forest is assigned an MSA value of 1.0 and a lightly used forest is set at 0.70 (i.e. a 30 percentage points loss in MSA). Now, by forcing a transition from one system state to another, for example by an ICIBs intervention to boost sustainable use, increase in MSA — or the prevention of MSA losses — can be estimated.

Making use of MSA values has several advantages and disadvantages. MSA values provide an effective and intuitive indicator of the biodiversity intactness in an area (Schipper et al., 2016). Moreover, the quantitative relationships between MSA values and land use as available in the GLOBIO model provide the opportunity to make estimations on the possible impacts of several management intervention categories as compared to the pristine state. However, limitations do exist herein as well: MSA values from GLOBIO v3.6 lack granularity of certain specific land use management categories and their corresponding MSA values, due to an absence or shortage of impact studies in the general literature (e.g. on mixed landscapes, organic farming or wildlife-friendly farming systems). Additionally, MSA alone is insufficient to capture all relevant aspects of biodiversity change. For example, because the undisturbed state of an ecosystem is used as a MSA reference scenario, 'rareness of species' as a distinct property of biodiversity is not part of the MSA indicator, while the assumption that human interventions always lead to a lower biodiversity value compared to a pristine situation is by definition included. Nevertheless, MSA values can serve as a first proxy for a decrease or increase in biodiversity in a system affected by changes in management (intensity). We discuss more about the use of MSA values in the discussion sections of the case studies and in Chapter 11.

Figure 2.2

Photographic impression of mean species abundance indicator at landscape level



Bron: PBL, 2009

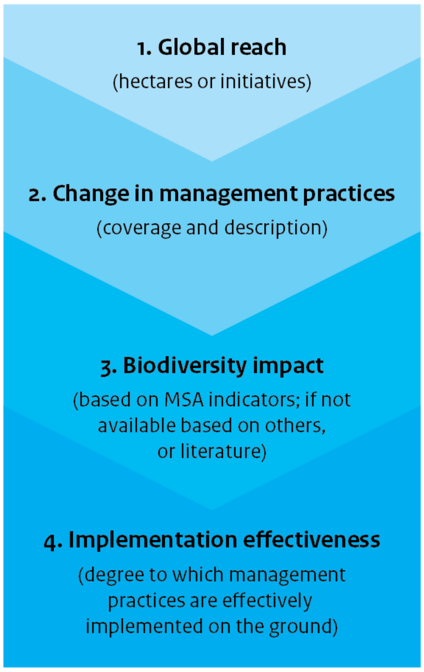
The value of the MSA indicator gradually changes due to human impacts from highly natural ecosystems (MSA of 90% to 100%) to highly cultivated or deteriorated ecosystems (MSA about 10% or less). Reprinted from 'Rethinking Global Biodiversity Strategies: Exploring structural changes in production and consumption to reduce biodiversity loss' (p. 38), by PBL Netherlands Environmental Assessment Agency, 2010, The Hague/Bilthoven.

2.4 Assessment framework

In order to systematically identify and aggregate the biodiversity benefits of ICIBs, we developed a framework, which is inspired by the work of Milder and colleagues (Milder et al., 2015). They proposed a three-level approach for the systematic evaluation of conservation impacts of sustainability standards in tropical agriculture. The approach ranges from system-wide monitoring (level 1), to sampled monitoring (level 2), to in-depth focused research (level 3), which verifies and calibrates level 1 and 2 monitoring.

We, however, adapted that model as depicted in Figure 2.3 below. It applies a similar logic as Milder et al., (2015) — going from system-wide monitoring to more in-depth assessments — and identifies indicators for outcomes and impacts of ICIBs at these different levels. This is based on combinations of: (1) descriptive and spatial data of the **global reach** of a specific ICIB, in terms of hectares and/or number and spread of projects around the world; (2) the identification of a **change in management practices** by a specific ICIB, compared to the pre-intervention situation, ideally including the area coverage of these different practices; and (3) an analysis of the expected **biodiversity impact** per management practice and its area coverage under level 2, and — if possible — the overall biodiversity impact of all these practices aggregated (based on MSA estimates and/or literature reviews of detailed cases). As it is highly likely that implementation challenges exist for each ICIB, we also aim — at a fourth level (4) — to provide an estimate of the **implementation effectiveness** of ICIB interventions, and to assess whether their projects are actually executed on the ground, so that their full potential, as indicated under levels 2 and 3, is indeed realised (based on more or less systematic reviews of field studies).

Figure 2.3
Assessment framework



While recognising that the outcome and impact of ICIBs can relate to both their own practices as well as to those of other actors, through for example, stimulation, inspiration, and catalysation of new actions by other agents, be it governments or companies, this study will focus on the actual biodiversity outcomes and impacts of ICIBs' own initiatives on the ground, and will not look at their broader governance functions and indirect effects (this is covered in related studies, see for example Pattberg et al., 2017; Kok et al., 2019).

At the first level of 'global reach', data refer to outcomes, for example the number of hectares of sustainably managed or conserved areas by ICIBs. These general outcome data are often available in, or can be retrieved from, existing literature or databases, or may be provided by the initiatives themselves. If these spatial data are not available, the number of projects of the specific ICIB around the world may suffice as well.

The second level of 'change in management practices' also involves outcomes, but more specific ones compared to level 1: the identification of specific practices, i.e. management interventions with potentially positive biodiversity impacts, as well as — ideally — their specific area coverages (i.e. a further specification of hectares as described in level 1 towards specific management practices; and if not available, further specification of level-1 information towards the number of ICIB projects for each specific practice).

At the third level of 'biodiversity impact', we aim to identify expected biodiversity gains of the specific management practices identified at level 2, as well as the overall, average biodiversity impact of all interventions combined. Here, we use MSA values corresponding with different land management types as included in the GLOBIO model (v3.6) (for details, see Section 2.3). In this study, in cases where area coverages of all different management practices of an ICIB are available, biodiversity gains are calculated by combining area coverages and MSA values into one indicator (in ha*MSA or ΔMSA·ha). Such an integrated indicator makes a comparison as well as an aggregation of different management practices in different area sizes possible. If data on area coverages are lacking, ΔMSA values are still indicated, but per management practice, implying that these cannot be aggregated. In cases where there are no matching MSA values available, insights from academic literature are used to provide at least some information on possible on-the-ground biodiversity gains of specific ICIBs.

Being aware that implementation often falls short of 100% effectiveness, the results of level 3 are put into perspective by assessing, where possible, to what degree the various management practices are actually executed on the ground. **Level 4 therefore provides an estimate of the 'implementation effectiveness'** of ICIBs, assessing the degree to which their full potential is being realised. These insights are retrieved from reviews of impact studies already published in the academic literature that address specific ICIBs (literature references to these reviews are provided in the respective case studies below).

We also conducted such literature reviews for cases that provided the possibility of more in-depth analyses ourselves. This resulted in four — more or less — systematic literature reviews: for forest certifications, for community forest management, for certification of agro-commodities and for organic agriculture. A full systematic literature review aims to identify, evaluate and integrate relevant findings available in the scientific literature to address a generic research question (Waddington et al., 2012). Such reviews systematically follow a number of procedural steps: collecting relevant literature, assessing the quality of data, in — and excluding studies based on quality, relevance, coverage and 'risk of bias', synthesising the various findings and generalising the overall impact. Such reviews are however very time consuming. For forest certification and community forest management, a full systematic review could be found through a master student's project, the results of which are published in a peer-reviewed Q1 journal; see Di Girolami et al., 2023). For the other two reviews — agro-commodities and organic agriculture — some of the authors of this report did the review work themselves, and for time-consuming reasons, could not apply all rules and procedures of systematic reviews. Yet they produced credible and usable overviews. For all details, see the case studies themselves.

2.5 Selection of initiatives

The final building block of this report's methodology is the case study approach. We employ a mixed method, evaluative, multiple case study design (Yin, 2009). The method is mixed since we combine qualitative and quantitative data, and the systematic literature reviews include qualitative and quantitative impact studies. It is evaluative, because we conduct an ex post impact assessment. And it is multiple, because we analyse multiple cases.

The selection of initiatives for this study is a three-step process (Table 2.1). First of all, a 'long list' of initiatives was created based on a quick scan of databases on voluntary sustainability standards by the International Institute for Sustainable Development (Potts et al., 2016), on climate, energy, forestry, fisheries, and agriculture initiatives (Pattberg et al., 2017) and on SDG 14 & 15 partnerships (SDG website), complemented by overviews of non-state initiatives in the literature (Selnes and Kamphorst, 2014; Roelfsema et al., 2018; Bennet et al., 2016). This analysis was further expanded through the snowballing method. The first 'quick scan' of the databases aimed at extracting possible biodiversity related, multi-stakeholder initiatives. Initiatives focusing only on protected areas were left out, since they do not categorise as Other Effective area-based Conservation Measures (OECMs) and are generally not considered international collaborative initiatives. Initiatives focusing only on single species were excluded because they focus on single species and do not necessarily aim to increase overall biodiversity.

Following this step, an in-depth website search was performed for each initiative to assess their mission statements, practices, goals, and governance structure. Initiatives suitable for this study were selected based on the following criteria, which resulted in a final longlist of 41 initiatives:

- Terrestrial, freshwater and wetland initiatives (marine initiatives were excluded due to time constraints and because the marine environment falls outside of PBL's current work scope);
- International multi-stakeholder initiatives with various types of actors in addition to solely public actors, with the intention to steer policy and the behaviour of their members or a broader community into a biodiversity friendly direction (so including multiple parties, covering actors from multiple countries and leaving out bilateral initiatives);
- A direct effect on biodiversity (broader governance functions, as recognised by Pattberg et al., (2017) such as norm setting and finance, are thus excluded).

To select for quantifiable and large-scale initiatives an initial assessment was performed through analysing websites and documents (e.g. annual reports) for size, data, activities and commitments. The final initiatives suitable for in-depth analysis:

- Are area-based and cover an area of substantial size (>100,000 ha), to have a potentially significant impact on global biodiversity;
- Provide clear commitments on concrete area-based measures for biodiversity conservation and sustainable use;
- Have data or impact studies available to quantify at least part of the biodiversity impacts;

- And preferably have minimal expected overlap with other initiatives.

This resulted in a shortlist of 21 initiatives.

To rule out missed initiatives due to outdated databases and articles, the list was checked by the projects' advisory board and an additional, separate, google search was performed using the keywords 'biodiversity', 'biodiversity initiative', 'biological diversity', 'biological diversity initiative', 'Convention on Biological Diversity', 'CBD', 'biodiversity project', 'international biodiversity project', 'nature', 'nature conservation', 'ecosystem services', and 'protected areas'. For each of these keywords, the search was limited to the first five search-result pages due to a continuing decrease in relevance. However, this search did not result in any additional initiatives for the list.

Due to a realisation that several relevant activities did not appear in our selection because of the absence of large, multi-stakeholder institutionalised initiatives, certain additional groups of activities were added as being ostensibly social movements. These are defined as follows: international, non-state movements that, compared to formal initiatives, are less institutionalised, often consisting of multiple decentralised groups (e.g. social media groups, online platforms, newsletters), but hold a similar sense of identity around certain perceptions on biodiversity or sustainable use, and these may thus have positive biodiversity outcomes, often organised around a specific approach/practice (e.g. community forestry, agro-ecology).

To come to a final selection from the shortlist, we discussed how many case studies would we have time for in our project (about 6 to 10) and decided to select these evenly spread across different themes (Table 2.1, column 1). The final selection of eight case studies is presented in Table 2.1, column 2. Note that some initiatives from our shortlist have been clustered into a single case study (see Table 2.1, column 3). Of this final selection, the following cases came from our search for social movements: Community Forest Management (CFM), Integrated Landscape Management (ILM), and Community Conservation (CC).

Table 2.1
Summary table of final case-study selection

Themes	Initiative/movement	Assessed example	Chapter number
Sustainable agriculture	VSS agro-commodities	Organic, RA, RSPO, FT, PF, GCP, UTZ, BP, RTRS, Global G.A.P	Chapter 3
	Organic agriculture	IFOAM	Chapter 4
Sustainable forestry	Community forest management	NA	Chapter 5
	VSS forestry	FSC & PEFC	Chapter 6
Sustainable landscapes	Integrated landscape management	Landscape initiatives in Africa	Chapter 7
	Greening cities	ICLEI	Chapter 8
Conservation Initiatives	Rewilding	Rewilding Europe	Chapter 9
	Community conservation	ICCA Registry	Chapter 10

CASE STUDIES

3 Agricultural Voluntary Sustainability Standards

Agriculture is one of the main drivers of terrestrial biodiversity loss, which is expected to be exacerbated with growing global population and changing consumption patterns (PBL Netherlands Environmental Assessment Agency, 2014). At present, 38% of land and 70% of freshwater consumption are allocated for agriculture (Food and Agricultural Organization, 2016a; World Bank Group, 2016). Despite its track record, the agricultural sector can also support and enhance biodiversity via practices such as agroforestry, integrated pest management, and planting perennial crops to provide pollinators and insects with habitats (De Beenhouwer, Aerts, and Honnay, 2013; TEEB, 2015). Clearly, numerous agricultural practices can be adopted to conserve and enhance on-farm and off-farm biodiversity, both for the sake of biodiversity itself as well as for the ecosystem services it delivers to the agricultural sector.

Agricultural Voluntary Sustainability Standards (AVSS) are voluntary schemes, which guide agricultural production towards sustainability in exchange for market recognition of sustainability standard-compliant production (Potts et al. 2016). Established to meet consumer demands for more sustainable products, AVSS have expanded significantly, representing or poised to represent close to 10% of total production in seven agricultural commodity sectors (bananas, coffee, cacao, palm oil, soya beans, sugar and tea) by 2020 (see Tables 3.1 and 3.2 for a summary of the global AVSS compliant area in seven focus crops and their biodiversity conservation production criteria). By virtue of promoting the adoption of more sustainable agricultural practices, AVSS have and will continue having an important role in protecting biodiversity in agricultural landscapes.

Nevertheless, determining the biodiversity conservation effects of implementing AVSS remains a challenge due to their inherent differences and the varying context in which they are implemented[3]. In addition to this challenge, AVSS focus on monitoring the adoption of agricultural practices as opposed to yielding tangible biodiversity impacts. AVSS also lack a standardised approach to measuring AVSS conservation impacts and have led to a limited number of rigorous and comparable AVSS biodiversity impact studies (DeFries, Fanzo, Mondal, Remans, and Wood, 2017)[4]. Yet, as AVSS continue to grow, there is an increased need to establish an evidence base for their impacts on biodiversity. This chapter aims to shed light on how AVSS are enabling biodiversity conservation by examining how they have been adopted by farmers over time and across the world across seven commodities (bananas, coffee, cacao, palm oil, soya beans, sugar, and tea) where prominent AVSS with international scope are most active, and how their implementation in the coffee and palm oil sectors are resulting in tangible biodiversity conservation impacts[5].

[3] For instance, Fair Trade originally focuses on social equity matters whilst Rain Forest focuses on the protection of ecosystems.

[4] De Fries et al. (2017) reviewed a total of 2600 peer-reviewed papers on the sustainability impacts of certification programs to find that only 2 out of 6 cases rigorously demonstrated real positive environmental impacts on the ground.

[5] The Organic standard is one of the most prominent standards operating in the seven crops where AVSS are the most active. Chapter 5 is solely dedicated to the Organic standard is also included in this report due to its wide

3.1 Motives, goals, targets

The main **motive** of AVSS is to enable more sustainable production and consumption of agricultural products by addressing the sustainability issues in the agricultural sectors they work in. The incentive to do so comes from existing and anticipated consumer demands for more sustainable agricultural products, sustaining agricultural production to maintain raw material supplies and lowering business reputational risks. As a sustainable development market-based instrument, AVSS aim to establish marketplace recognition for sustainable farming practices which can result in improved prices and market access for AVSS compliant farmers. In this way, AVSS are linking consumers and producers to enable sustainable development. Preventing agriculture driven environmental degradation such as deforestation, greenhouse gas emissions, and water depletion and pollution, all of which are resulting in biodiversity losses and undermining agricultural long-term productivity, has also been an important motive for establishing AVSS.

To enable more sustainable agriculture, AVSS have their own visions, missions, goals, objectives and theories of change, which can collectively be referred to as their **goals**. For instance, Fairtrade aims to 'promote sustainable development and to reduce poverty through fairer trade' while the Rainforest Alliance aims to 'conserve biodiversity and ensure sustainable livelihoods by transforming land-use practices, business practices and consumer behaviour' (Fairtrade International, 2011; Rainforest Alliance, 2017).

The AVSS reviewed for this chapter tend to establish **targets** related to increasing their membership and production volumes as opposed to setting sustainability impact targets. For instance, Bonsucro — an international non-profit multistakeholder governance group to promote sustainable sugarcane — aimed to attain 150 members and a 20% market penetration by 2017 (Bonsucro, 2016). Some standards convey targets via their visions to transform the sectors they work in. For instance, the Roundtable on Sustainable Palm Oil's (RSPO) vision is 'sustainable palm oil is the norm' (RSPO, 2018, p.9). Due to the focus of AVSS on enabling more sustainable modes of agricultural production, targets to expand their presence and uptake in the sectors they work in, aligns with many of the CBD Kunming-Montreal COP15 Targets, particularly Target 10, 11, and 12 which aim to prevent natural habitat loss, enable sustainable agriculture to conserve biodiversity and lower pollution to non-detrimental levels for ecosystems and biodiversity.

3.2 Theory of change: from input to impact

AVSS are supply chain governance systems that respond to consumer and market demand for sustainable production[6]. The Impact, Output, Outcome, and Impact (IOOI) model provides a structured way to examine the effects of AVSS on biodiversity conservation. Consumer and market demand for sustainable products to address the pervasive effects of production systems constitute the **Input**. Consumers may be willing to pay more for assurance that AVSS-compliant products are

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proliferation and prominence across the agricultural sector, which goes well beyond the seven crops covered in this case study.

[6] By recognizing environmental and social values along with cost and quality, the social demand transforms a commodity market into a siloed differentiation of products — e.g. coffee, tea, sugar or palm oil — and pushes towards the adoption – formal or tacit – of production standards, where international companies want to avoid being identified with below-standard practices.

supporting transformation towards more sustainable product value chains. AVSS organisations typically consist of supply chain stakeholders (e.g. farmers, traders, brands) working towards achieving sustainability goals, reflected in their principles, criteria and indicators are considered **Outputs (Table 3.1)**[7]. This relates to how the goals of AVSS are operationalised via their sustainable production criteria, which are periodically reviewed to remain current and relevant[8]. **Table 3.1** provides a cursory examination of AVSS production criteria with direct implications on biodiversity across the seven focus crops examined[9]. It shows that AVSS possess differing levels of importance for conservation of protected areas, areas with high conservation value, and on-farm natural habitats. Whereas, water and soil conservation as well as reducing pollution from agrochemical use are equally important across the AVSS. The production criteria implementation requirements can also vary significantly. For instance, organic production of crops prohibits the use of synthetic pesticides while all the other AVSS do not make this imperative (Potts et al., 2016). Another major intra-AVSS difference that becomes visible from Table 3.1, is between their assurance system, even though the majority of the AVSS examined require third-party certification. These differences translate into varying potential for AVSS to conserve biodiversity which is also greatly influenced by the diversity of farming contexts in which they are being applied. The **Outcomes** consist of the uptake and adherence to AVSS by supply chain stakeholders aiming to enable more sustainable agricultural production. The **Impacts** are the actual tangible biodiversity conservation effects of implementing AVSS.

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[7] AVSS principles include economic, social and environmental sustainability aspects that aim to improve producer livelihoods and incentivize more sustainable production practices.

[8] Sustainable production standards are often reviewed every 5 years.

[9] Bananas: Rainforest Alliance, Fairtrade, Organic, GlobalG.A.P., - Cocoa: Rainforest Alliance, UTZ Certified, Fairtrade, Organic - Coffee: Global Coffee Platform, Rainforest Alliance, UTZ Certified, Fairtrade, Organic - Palm Oil: Roundtable on Sustainable Palm Oil, Rainforest Alliance, Organic - Soya beans: Roundtable on Responsible Soy, Proterra Foundation, Organic, Rainforest Alliance - Sugarcane: Bonsucro Platform, Rainforest Alliance, Fairtrade, Organic - Tea: Rainforest Alliance, UTZ Certified, Fairtrade, Organic

Table 3.1
Alignment between biodiversity friendly agricultural practices and production criteria of agricultural voluntary sustainability standards.

VSS[10]	Focus crop covered	Assurance system	Avoid Impact on Protected and HCV areas	Avoid impact on non-protected natural habitats	Provide on-farm habitat	Reduce impacts of invasive species	Reduce pollution from agro-chemical use	Water conservation	Soil conservation
	All 7 Crops	Certification	•	-	•	-	•	•	•
	Bananas, Cocoa, Coffee, Oil Palm, Tea	Certification	•	•	•	•	•	•	•
	Oil Palm	Certification	•	-	•	•	•	•	•
	Bananas, Cocoa, Coffee, Cotton, Sugarcane, Tea	Certification	•	-	•	•	•	•	•
	Soya beans	Certification	•	•	•	•	•	•	•
	Coffee	Verification	•	•	•	-	•	•	•
	Cocoa, Coffee, Tea	Certification	•	•	•	-	•	•	•
	Sugarcane	Certification	•	□	•	-	•	•	•
	Soya bean	Certification	•	•	•	•	•	•	•
	Bananas	Certification	•	□	□	•	•	•	•

Alignment between biodiversity friendly agricultural practices and the required (filled circles), partial (open squares) or absent production criteria of agricultural voluntary sustainability standards operating in the seven focus crops (bananas, cocoa, coffee, oil pail, soya beans, sugarcane, tea). The information was obtained and modified from Tayleur et al., (2016) and Potts et al., (2016).

[10] RA=Rainforest Alliance, RSPO = Roundtable on Sustainable Palm Oil, FT = Fairtrade International, PF = Proterra Foundation, GCP = Global Coffee Platform, UTZ = UTZ Certified, BP = Bonsucro Platform, RTRS = Roundtable on Responsible Soy, GG = Global G.A.P. The baseline before the land became AVSS certified can be assumed to be conventional farming.

3.3 Past performance: assessing outcome and impact

Agricultural certification schemes represent an institutionalised group of ICIBs. Therefore it seems that it could be rather straightforward to assess their biodiversity impacts through the different levels of the assessment framework. In terms of global reach, numbers seem to be clear. For level 2 a slightly different approach is needed for assessing the impact of AVSS. In the end, this led to an assessment up to level 1.

3.3.1 Level 1: Global reach

The focus crops of this chapter are mostly grown in the tropical regions of the world that contain important biodiversity hotspots, where primary forest and grassland ecosystems are still present in relatively large amounts (Milder et al., 2015). For this reason, expanding AVSS-compliant production within these crops represents an important biodiversity conservation opportunity. In 2015, the agricultural lands supporting AVSS-compliant production across the seven focus crops (bananas, coffee, cocoa, palm oil, soya beans, sugar, and tea) expanded significantly to occupy **between 12 and 16 million hectares** making up 5.9% to 8.1% of the land dedicated to these crops (or between 0.7% and 0.9% of global crop land) (Lernoud et al., 2017). Table 3.2 conveys the total agricultural land dedicated to AVSS-compliant production in 2015 within the banana, cocoa, coffee, palm oil, soya bean, sugar, and tea sectors. Except for sugarcane and soya beans, more than 10% of the total land dedicated to these crops supported AVSS-compliant production in 2015. In that year, the largest production areas for AVSS-compliant bananas, coffee, soya beans, and sugarcane were found in South America, the largest areas dedicated to AVSS-compliant palm oil and tea were found in Asia, and the largest AVSS-compliant cocoa production areas were located in Africa. The Cumulative Annual Growth Rate (CAGR) between 2008 and 2015 of AVSS-compliant production area conveys how it has grown over time. All sectors experienced CAGR 2008–2015 of more than double digits with the cocoa, oil palm, and sugarcane sectors experiencing noticeably higher growth rates. Please note that all CAGR 2008–2015 values calculated and presented in Table 3.2 are based on minimum AVSS-compliant production area estimates.

Table 3.2
Harvested area of the seven major crops where AVSS were most prominent in 2015

	Sugarcane	Palm oil	Tea	Soya beans	Coffee	Cocoa	Banana
Area 2015 (ha)							
Conventional	25,477,565	17,393,488	3,204,068	117,718,089	6,136,161	6,823,077	4,995,571
	(95.55%)	(86.02%)	(81.71%)	(97.46%)	(56.92%)	(68.72%)	(91.72%)
AVSS-Compliant	1,140,129	2,783,615	559,692	2,552,243	2,611,969	1,809,200	290,544
	(4.28%)	(13.77%)	(14.27%)	(2.11%)	(24.23%)	(18.22%)	(5.33%)
Pot. AVSS-Compliant	46,717	44,353	157,576	521,857	2,031,905	1,296,375	160,537
	(0.18%)	(0.22%)	(4.02%)	(0.43%)	(18.85%)	(13.06%)	(2.95%)
Total	26,664,411 (100.00%)	20,221,456 (100.00%)	3,921,336 (100.00%)	120,792,189 (100.00%)	10,780,035 (100.00%)	9,928,652 (100.00%)	5,446,652 (100.00%)
CAGR 2008–2015							
	Sugarcane	**Palm Oil**	**Tea**	**Soya beans**	**Coffee**	**Cocoa**	**Banana**
Conventional	0.94%	2.22%	1.66%	3.63%	-7.58%	-5.22%	-0.20%
AVSS-Compliant	65.84%	72.83%	37.24%	11.32%	26.45%	51.17%	17.34%
Total	1.67%	4.70%	4.65%	3.63%	0.31%	0.63%	0.88%

Conventional stands for non-AVSS-compliant; VSS-compliant stands for area that complies with at least one AVSS; Pot. VSS-compliant stands for area whose allocation to conventional or to AVSS cannot be clearly ascertained based on available data; VSS Comp. Max stands for the sum of the area reported by all AVSS; and CAGR stands for Cumulative Annual Growth Rate. Source: elaborated by the author, based on statistics of the Food and Agricultural Organization (FAO) and the AVSS applicable to the seven crops reviewed.

3.3.2 Level 2: Change in management practices

AVSS in the seven focus crops have significantly expanded in production volumes across time and geographies. The expansion of AVSS production presented in **Table 3.2** across seven focus crops along with AVSS production criteria summarised in **Table 3.1** provide insights into their potential for having positive biodiversity impact. Although their rapid and widespread uptake is a favourable outcome for biodiversity, their actual effectiveness can only be determined by assessing their biodiversity impacts that were implemented. Such an assessment is based on assessing the impact and coverage of the practices related to certification such as the protection of natural sites and conserving water and soil resources. With such management interventions, or rather non-interventions, AVSS could significantly increase the naturalness, or MSA value, of an area compared to conventional farming.

To assess this in level 2, we would need the coverage per management practice to later connect it to an MSA category. Unfortunately, coverage of the specific production criteria in hectares is not available. Furthermore, it will be difficult to assess some of these criteria in hectares as well. For this information to be available, certificates would need clear monitoring reports, or for the information to become available in the future with the advancement of remote sensing technologies. Another possibility would be to create a general MSA value for certified cropland, thereby assessing all certification schemes with one average impact level. To give an example, in level 3 we could couple this with the categories provided by Alkemade et al., (2009). Here, low-input agriculture is represented by an MSA value of 0.3, while intensive agriculture is represented by an MSA value of 0.1. Which would mean that the total AVSS certified land of the crops from this case study (12–16 million hectares), undergoes an MSA increase of 0.2. The reason why we do not do this here, is because low-input agriculture is described as 'Subsistence and traditional farming, extensive farming, and low external input agriculture', which generally does not apply to AVSS-certified crops. For that, we would need a new MSA category. Nevertheless, this gives an example of how this MSA value increase could be calculated.

To avoid generalising all crops and certification types, it could be useful to make a division in level 2 in crop types or certification schemes. As can be read in the following chapter on **'organic agricultural certification'**, organic certification can have a different impact on biodiversity depending on the crop type. This could also be the case for different crop types under different certification schemes. In that case, the total area per crop type or certification scheme could be gathered for level 2 and used to calculate level 3 and 4. This does mean that for level 3 and 4 MSA values have to be available for crop types or certification types.

3.3.3 Level 3: Biodiversity Impacts

Only case studies are available that are mostly crop- and AVSS-focused and specific, and therefore not generalisable. However, looking at specific practices also used in AVSS certification can provide insights into the biodiversity impacts of AVSS.

Impact studies on AVSS operating in the coffee and palm oil sectors are examined to determine their overall biodiversity impact. The coffee and oil palm sectors were chosen due to the long-standing presence of AVSS in the coffee sector and recent attention on deforestation associated with the expansion of oil palm plantations. The impact studies reviewed for both sectors point to a mix of results in the effectiveness of AVSS to conserve biodiversity. DeFries et al. (2017) indicate that there is a dearth of rigorous and comparable studies examining the sustainability impacts of AVSS programmes and that not all their interventions lead to significant positive environmental outcomes. Nevertheless, some of the mounting scientific evidence of AVSS programmes impacts on biodiversity is presented pertaining to biodiversity conservation.

Coffee is a perennial crop that can be cultivated in agroforestry and full sun settings. Agroforestry coffee supports significantly more biodiversity than full sun coffee plantations. A meta-analysis conducted in Latin America, Africa, and Asia concluded that converting natural forests to agroforests and converting agroforests into plantations with sparse shade tree leads to a decline of 11% and 46% in species richness, respectively (De Beenhouwer et al., 2013). The study concludes that cultivation intensification and species richness decline follow a concave yield function. Consequently, AVSS that require coffee to be grown in an agroforestry setting with shade requirements are well positioned to conserve biodiversity. In addition to shade requirements, AVSS in the coffee sector with production criteria requiring the conservation of protected and non-protected natural habitats are also likely to have tangible biodiversity conservation impacts when properly implemented.

A meta study conducted by the Committee on Sustainability Assessments (COSA) (2013) examined and compared standard-compliant and conventional coffee and cocoa farms in 12 countries (6 Latin American, 3 African, and 3 Asian countries) by analysing 18,000 survey examining Organic, Fairtrade, Rainforest Alliance, UTZ Certified, CAFÉ Principles, Nespresso AAA, and 4C standards. Their overall results indicated that coffee and cocoa farmers adhering to a AVSS programme were more likely to practise soil and water conservation measures and support biodiversity compared to conventional farmers. COSA points out that their overall results can vary significantly per country and AVSS. Haggar et al. (2017) compares certified (CAFÉ Practices, Fairtrade, Organic & Fairtrade, Rainforest Alliance and UTZ Certified) versus non-certified coffee farms in Nicaragua and finds that the certified farms performed better in tree density, percentage of farms with 3 tree strata, above ground carbon, percentage of plant ground cover, and — except for CAFÉ Practices — tree basal area. Vanderhaegen et al., (2018) examined 74 coffee farms with and without multiple AVSS in Mount Elgon National Park in Uganda. They found that Fairtrade & Organic coffee farms performed better in terms of carbon storage, tree diversity and density and entomofauna abundance and diversity compared to conventional coffee farms while 4C, Rainforest Alliance & UTZ coffee farms performed worse on the same indicators than conventional farms. Takahashi & Todo, (2013 and 2017) observed in Ethiopia that Rainforest Alliance certification for smallholders has a marginal and statistically insignificant positive impact of 1.7% on forest conservation but contributes to halting forest degradation within 100 metres around farms which appears to be motivated by farmers being able to sell their product at 15% to 20% higher prices[11].

With vastly superior yields compared to other oil crops[12], growing global demand for palm oil[13] has led to the conversion of large tropical rainforests to give way to oil palm plantations[14]. Large-scale conversion of rainforest to oil palm plantations is one of the most important causes of biodiversity declines in Southeast Asia, particularly in Indonesia and Malaysia (Lucey et al., 2014a). From 1990 to 2010, around 17% and 63% of new plantations in Malaysia and Indonesia, respectively, took place in tropical forests, and up to 30% occurred on peat soils resulting in natural habitat losses, biodiversity declines, increased wildfires and CO_2 emissions (Pirker et al., 2016)[15]. Much like the coffee sector, oil palm plantations that support diverse vegetation (i.e. multi-storey vegetation and ground cover) are more likely to conserve biodiversity. On average, 5 arthropod orders were measured for polyculture oil palm farming versus 4 for monoculture oil palm farming in 120 sites in peninsular Malaysia (Ghazali et al., 2016). Forest patches within plantations may help conserve biodiversity conservation and control pests in oil palm plantations in Sabah, Malaysia (Lucey and Hill, 2012; Lucey et al., 2014b). However, Edwards et al. (2010) makes the case that protecting contiguous forests is far better for protecting biodiversity than retaining forest fragments as they found that imperilled bird species in forest fragments and oil palm plantations located in the Ulu Segama-Malua forest reserve in Sabah, Malaysia were 60 and 200 times lower than contiguous forests respectively (Edwards et al., 2010).

A number of AVSS active in the sector have emerged to try and curb the devastating impacts that the palm oil sector has had on tropical deforestation and biodiversity. The Roundtable on Sustainable Palm Oil (RSPO) is the most prominent AVSS in the palm oil sector. Its implementation in Kalimantan and Sumatra, Indonesia was found to reduce deforestation rates of close to 10% per year in non-certified plantations down to 7% per year for RSPO-certified plantations between 2001 and 2015 resulting in a 33% deforestation reduction in RSPO-certified oil palm plantations (Carlson

[11] According to Takahashi & Todo (2013) conservation seems to be related to smallholder ownership rather than certification.

[12] Oil palm gives 6 to 10 times more vegetable oil per hectare than the other major oil crops such as rapeseed, sunflower and soya bean (Godswill et al., 2016)

[13] Palm oil became the most consumed edible oil in 2010 (MarketsAndMarkets, 2017). It is coming under increasing scrutiny as its ideal growing environment intersects with highly biodiverse lowland tropical rainforests located below 500 metres in altitude 10° North and South of the equator (Goh, Wong, & Ng, 2017).

[14] Vijay et al. (2016) examine the loss of tropical forests due to oil palm plantations and in Southeast Asia, South America, Mesoamerica, and Africa, and find that 45%, 31%, 2% and 7% respectively of the sampled areas in these continents were tropical rainforests in 1989.

[15] In addition to being biodiversity reservoirs, peatlands store approximately 528,000 million tons C, one-third of the global soil carbon (Khatun, Reza, Moniruzzaman, & Yaakob, 2017).

et al., 2018). No differences were found in fire detections from 1999 and 2015 and Orangutan population decline from 2009 to 2014 between RSPO-certified and non-certified plantations in Kalimantan, Indonesia (Morgans et al., 2018). Carlson et al. (2018) point out that there is the risk that certification incentivises deforestation, since the 97% of the deforestation they observed in Kalimantan and Sumatra, Indonesia occurred pre-certification, perhaps to get ahead of being restricted to expand operations into areas with more conservation value.

The findings presented above focused on examining the role of AVSS in protecting natural habitats providing a limited picture on their biodiversity conservation potential. Their implementation can result in soil and water conservation as well as greenhouse gas emission and agrochemical input reductions all of which also support biodiversity (Committee on Sustainability Assessments, 2013). Examining the biodiversity conservation impacts of individual agricultural practices could provide a way to overcome the lack of AVSS and biodiversity impact studies. For instance, a global meta-analysis study on restoring riparian habitats reports that it enhances biodiversity by an average of 68% based on response ratios to restoration interventions (Barral, Rey Benayas, Meli, and Maceira, 2015). Other global meta-analysis studies report that crop rotation improves soil microbial richness and diversity by 15% and 3%, respectively, and that no-till improves soil microbial carbon biomass by at least 25% (Li, Chang, Tian, and Zhang, 2018; Venter, Jacobs, and Hawkins, 2016). A global study on soil conservation techniques finds that they reduce soil losses by 88% and water runoff by 53% (Xiong, Sun, and Chen, 2018). Although impact studies on farming practices are no substitute for rigorous and comparable AVSS and biodiversity impact studies, they can provide insights on AVSS biodiversity conservation if their production criteria require their farmers to adopt them.

The studies summarised above provide some evidence within the coffee and palm oil sectors that AVSS are positively impacting biodiversity in specific contexts. From these studies, mean species abundance difference estimates could be derived for the specific contexts in which AVSS are being applied, providing a step towards establishing their overall biodiversity impact.

3.3.4 Implementation effectiveness

AVSS are useful albeit indirect interventions that contribute towards biodiversity conservation. However, due to the inherent structure of AVSS as a standard, they doesn't directly contribute towards biodiversity increase. Moreover, the contribution as well has often been contested within scholastic arenas. This report identifies AVSS as an ICIB that has a crucial role to play towards biodiversity conservation. However, it cannot be sure of the implementation effectiveness of AVSS. The reasons for the lack of an ability to measure implementation effectiveness for AVSS are as follows. Firstly, as AVSS are sustainability and voluntary standards to be adopted by farmers, most studies have focused on farmer compliance towards the AVSS. This report therefore encourages research to be undertaken in measuring the exact on-the-ground compliance of AVSS. Secondly, as AVSS comprise too many crops, it is difficult to identify the exact implementation effectiveness for each crop. In order to address this, this report narrowed down its focus on coffee and palm plantations. Even so, due to the multiplicity of AVSS in different parts of the world, a conclusive measure was not feasible within the scope of this study. Thirdly, the limited results that this study could collate on the implementation effectiveness of AVSS was inconclusive and varying. This variability in results is further discussed in the next section and also qualifies the lack of confirmation of implementation effectiveness.

3.4 Discussion and conclusion

AVSS have already had quite a significant global reach, which can be expected to grow with the current global focus on creating more sustainable agricultural systems. The agricultural lands supporting AVSS-compliant production across the seven focus crops (bananas, coffee, cocoa, palm oil, soya beans, sugar, and tea) have expanded significantly to occupy between 12 and 16 million hectares making up 5.9% to 8.1% of the land dedicated to these crops (or between 0.7% and 0.9% of global crop land) in 2015 (Lernoud et al., 2017), thereby providing an important opportunity for increasing biodiversity.

Assessing how AVSS are protecting biodiversity is complicated by the fact that they are not created equal — they have their own theories of change, production criteria, assurance systems and governance models which align with the different crops and geographies they work in (see Table 3.1). At the outset, AVSS can be expected to have varying effects on biodiversity conservation. Furthermore, AVSS are implemented in various contexts where they can have positive, negative, or neutral effects on biodiversity, calling into question the general assumption that their implementation inevitably leads to additional biodiversity conservation. In many cases, crop variety and cultivation approach will have a more profound impact on biodiversity than the adoption of AVSS-compliant production criteria. For instance, agroforestry grown coffee will likely support more biodiversity than AVSS-compliant coffee grown in full sun plantations (De Beenhouwer et al., 2013). Of course, AVSS with shade requirements encourage agroforestry coffee cultivation. In cases where farmers are already implementing farming practices aligned with AVSS production criteria their adoption will likely not result in significant additional biodiversity conservation benefits. These context specific considerations require comparable and rigorous research to determine if AVSS are resulting in additional biodiversity conservation benefits.

At this point, no generalisable information is available on the biodiversity impacts of AVSS in the sectors we examined. As a baseline for the AVSS-certified land, we assume that pre-certification the practices were in the category of conventional farming. Conventional farming has a corresponding MSA value of 0.1 (Alkemade et al., 2009). However, there is no MSA category yet available for AVSS-certified farming, due to a lack of research on the impacts of AVSS practices on biodiversity. Therefore, calculating the increase in the MSA value due to certified farming, let alone over different crop types or schemes, is not possible. There is an MSA category available for low-input farming — 'Subsistence and traditional farming, extensive farming, and low external input agriculture', which corresponds to an MSA value of 0.3 (Alkemade et al., 2009). This does not, however, directly relate to the practices of AVSS since they represent a less intensive way of farming. When more information comes available on the biodiversity impacts of AVSS practices for different crop types and continents, corresponding MSA categories could be created for AVSS.

Conclusive evidence on the effects of AVSS on biodiversity conservation is challenged by a lack of rigorous and comparable impact studies preventing an overall assessment of AVSS biodiversity benefits (Komives et al., 2018). To further support this finding, a recent search on the ISEAL Evidensia, an online repository of information on AVSS impacts, yields 24 results/studies when searching for biodiversity (ISEAL, WWF, and Rainforest Alliance, 2019). Some studies have found no significant differences between AVSS-compliant and conventional agricultural production practices pertaining to the protection of the environment and biodiversity but even these are limited in terms of comparability, rigour, AVSS-covered, agricultural commodity and geographical scope (DeFries et al., 2017). Due to this current dearth of information, establishing the implementation gap for AVSS to reach their full potential in conserving biodiversity is not possible at this moment.

Efforts to establish a stronger evidence base for AVSS to understand how they are impacting biodiversity are ongoing by moving towards monitoring impacts as opposed to the adoption of farming practices (Komives et al., 2018; Milder et al., 2015). This important shift would allow AVSS to better tailor and optimise production criteria where they are applied for biodiversity conservation. Landscape approaches to AVSS-compliant production are also being contemplated which would allow for more holistic approaches to conserving biodiversity (Mallet et al., 2016; Potts et al., 2016)[16]. In addition to impact studies, better data on understanding how AVSS-compliant products flow across the globe would allow governments to design more effective sustainable agricultural production policies potentially enabled by AVSS. As developed for the Organic standard, HS (Harmonised System, or product) codes for other AVSS-compliant products are sorely required to better track their exports and imports (Potts et al., 2016).

To conclude, consumer **input** demanding more sustainable agricultural products has resulted in the emergence of AVSS as one of its **outputs**. AVSS-compliant production is an **outcome**, which should result in biodiversity conservation but since standard compliance is for the most part, based on the adoption of agricultural practices their tangible sustainability **impacts**, including those on biodiversity remains deficient. Despite the need for more comparable and rigorous AVSS impact data, they have expanded significantly in the last two decades growing at a CAGR 2008–2015 of at least 10% across bananas, cacao, coffee, palm oil, soya beans, sugar, and tea sectors, with presence in the majority of the focus crop growing continents (see Table 3.2). Despite this expansion, conclusive and generalisable evidence that they are resulting in improved biodiversity conservation has not yet been reached for all contexts in which they are applied (DeFries et al., 2017). The AVSS biodiversity impact studies reviewed for the coffee and palm oil sectors provided mixed results and expanding AVSS in these sectors will greatly benefit from rigorous impact monitoring. Whether AVSS continue to expand or not, they have laid important groundwork towards enabling sustainable consumption and production via their supply chain governance systems and farming production criteria allowing consumers to support, through their purchasing decisions, the embodiment of sustainability and biodiversity conservation in agriculture.

[16] A landscape approach to implementing AVSS would be partly enabled by remote sensing technologies which are lowering large-scale monitoring costs (Tscharntke et al., 2015).

4 Organic Agriculture Certification

The International Federation of Organic Agriculture Movements (IFOAM – Organics International) is an umbrella organisation that groups together all organic agriculture standard bodies across the world, consisting of a diversity of binding systems established at different geographic scales. IFOAM – Organics International plays a fundamental role in reporting the state of organic agriculture. The 2018 World of Organic Agriculture report on the state of certified organic agriculture included data from 178 countries where formal certification and national reporting on organic agriculture were conducted (Willer and Lernoud, 2018)[17]. These organic agriculture certification systems comply with the IFOAM – Organics International standard benchmark: 'The IFOAM – Organics International Family of Standards contains all standards officially endorsed as organic by the Organic Movement, based on their equivalence with the Common Objectives and Requirements of Organic Standards. Both private standards and government regulations are admissible (Willer and Lernoud, 2018)[18].'

The IFOAM – Organics International family of standards are referred to in this report as 'the organic agricultural standard', which has become one of the most widespread Agricultural Voluntary Sustainability Standards (AVSS) in the world. In 2016, farming compliant with this standard occurred across 57.82 million hectares of agricultural and grass land or approximately 1.2% of total global cropland (Willer and Lernoud, 2018)[19]. The organic standard establishes rules, from voluntary guidelines to binding regulations, for more sustainable agricultural production methods along entire agricultural commodity chains allowing for end-product differentiation (IFOAM, 2014).

Organic agriculture was greatly motivated by Rachel Carson's 'Silent Spring' which documented the biodiversity impacts of agrochemical use in 1962 (Carson, 1962). Ever since, organic farming has focused on reducing and eliminating agrochemicals (synthetic fertilisers and pesticides), which have been used to intensify farming throughout the 20th century and resulted in significant negative environmental impacts and biodiversity losses. The rise of organic agriculture movements called for ('Definition of Organic Agriculture | IFOAM' n.d.) coordinated action which led to the foundation of the IFOAM in 1972. Participation in organic certified agriculture is driven in part by the growing need to protect natural environments and biodiversity from unsustainable agricultural practices: 'Organic Agriculture is a production system that sustains the health of soils, ecosystems and people. It relies on ecological processes, biodiversity and cycles adapted to local conditions, rather than the use of inputs with adverse effects. Organic Agriculture combines tradition, innovation and science to benefit the shared environment and promote fair relationships and a good quality of life for all

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[17] In IFOAM reporting "organic agriculture land" includes land "in-conversion" which refers to organically managed land transitioning from conventional agriculture towards certified organic agriculture.

[18] A list of national organic agriculture standards can be found at www.ifoam.bio/orgs.

[19] From the 57.82 million hectares allocated to organic agriculture in 2016, 37.96 million were permanent grasslands, 10.61 million hectares were arable cropland, and 4.54 million hectares were permanent cropland (IFOAM – Organics International reports 4.26 million hectares of organic land whose agricultural use was unclear in national reports). The organic standard is also applied to wild collection areas, beekeeping, aquaculture, as well as forests and grazing areas outside agricultural land adding up 39.7 million hectares reported in 2016. However, the areas dedicated to these organic practices are not consistently reported by the countries that support them (Willer & Lernoud, 2018).

involved' (IFOAM, 2008).

Approximately 60% of human nutrition comes from rice, maize and wheat which were grown on approximately 43% of the world's total agricultural land in 2016 (Food and Agriculture Organization, 1995; FAOSTAT, 2016) and given their nutritional importance, this number is not likely to change. Very little agricultural land dedicated to these staple crops (approximately 1%) supports standard-compliant production (Potts et al., 2016; Tayleur et al., 2016). The organic standard is the most prominent AVSS operating in the staple crops and is strategically positioned to support the expansion of more sustainable agricultural practices in these sectors to protect biodiversity (Potts et al., 2016). One fifth of all organic agricultural arable and permanent cropland (10.33 million hectares) was allocated to the three main staple crops in 2015 (or approximately 2.25 million hectares-1.56 million hectares were allocated to wheat; 0.44 million hectares to maize, and 0.25 million hectares to rice) (FiBL statistics, 2019). Furthermore, certification of these crops presents great opportunities for conservation not only due to their large coverage, but also because of their large impacts. Wheat, for example, is one of the most important sources for biodiversity loss in agriculture (Potts et al., 2016) and wet rice cultivation the largest greenhouse gas emitter in South and Southeast Asia after deforestation (Ellis, 2014; Kritee et al., 2018; van Groenigen, van Kessel, and Hungate, 2013).

Although organic staple crop production focuses on fostering biodiverse agroecosystems, assessing its biodiversity impacts is challenging due to the varying contexts in which it is applied and availability of comparable impact studies (Barbieri, Pellerin, and Nesme, 2017). This chapter examines the role of the organic standard in conserving biodiversity in the staple crops by looking at its market presence and practices within the wheat, maize, and rice sectors, and reviewing organic agriculture biodiversity impact studies[20].

4.1 Motives, goals, targets

The main motive of the organic movement is the 'broad adoption of truly sustainable agriculture, value chains and consumption in line with the principles of organic agriculture' (IFOAM 2017, 2). This implies adopting farming practices yielding near agrochemical-free products that are less harmful to natural environments and human health[21]. To this end, the organic standard limits agrochemicals and bans the most harmful ones. Organic farmers who practise agrochemical-free agriculture must rely on biodiversity to improve their crop resilience to local growing conditions, pests and diseases and adopt farming practices that secure long-term soil fertility. Organic agriculture requires premiums to maintain its economic viability due to its lower yields and certification process (Seufert, 2019). For this reason, consumer campaigning constitutes one of the pillars of IFOAM – Organics International's activities (IFOAM, 2017).

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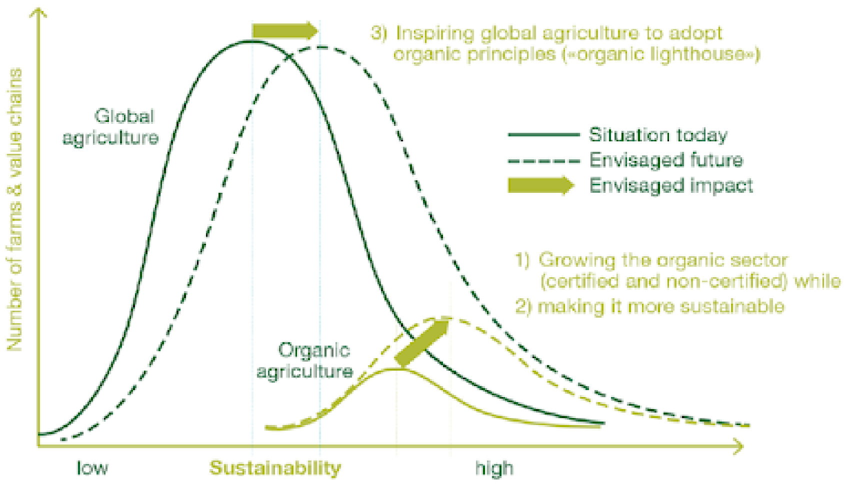
[20] The organic standard is also examined in the AVSS and Biodiversity case study within the bananas, cocoa, coffee, palm oil, soya beans, sugar and tea sectors. This case study focuses on its presence within the staple crops since it is the most prominent AVSS operating in these sectors.

[21] Gomiero (2018) reports that organic food is less likely to be contaminated with pesticides compared to conventionally grown food but that there are no major differences between the two farming systems concerning heavy metals, mycotoxins and bacterial contamination.

The IFOAM – Organic International 2017–2025 strategic plan consists of the following two goals: 1) 'increase the uptake of organic agriculture and similar approaches, certified or non-certified' and 2) 'increase the number of agriculture operations that are becoming more sustainable and integrate organic principles and methods' (IFOAM 2017, 2). These goals are reflected in Figure 4.1, representing the intent to increase organic farming while contributing to moving the agricultural sector towards more sustainability.

Figure 4.1
'Our goal for better and more organic agriculture and for a more sustainable global agriculture (IFOAM, 2017).'



The current targets of the global organic movement involve increasing the rate of conversion to organic agriculture, reducing the yield gap between organic and conventional farming, rewarding the delivery of common goods such as ecosystem services and obtaining recognition in government policies, since these tend to support only conventional production (Arbenz, Gould, and Stopes, 2016, 8). Due to the focus of the organic standard on enabling more sustainable modes of agricultural production, targets to expand their presence and uptake in agriculture align with the Kunming-Montreal CBD COP15 Targets 10, 11, and 12 which aim to prevent natural habitat loss, enable sustainable agriculture to conserve biodiversity and lower pollution to non-detrimental levels for ecosystems and biodiversity.

4.2 Theory of change: from input to impact

The organic standard is a supply chain governance system that responds to consumer and market demand for sustainable production[22]. The Impact, Output, Outcome, and Impact (IOOI) model provides a structured way to examine the effects of the organic standard on biodiversity conservation. Consumer and market demand for sustainable products to address the pervasive effects of production systems constitute the **Input**. Consumers willing to pay more for organic

[22] By recognizing environmental and social values along with cost and quality, the social demand transforms a commodity market into a siloed differentiation of products and pushes towards the adoption – formal or tacit – of production standards, where companies want to avoid being identified with below-standard practices.

certified products that are more sustainable (Arbenz, Gould, and Stopes, 2016). Organic organisations comprised of supply chain stakeholders (farmers, traders, manufacturers, retailers, and NGOs) working towards achieving sustainability goals, reflected in the production criteria of their standards are considered **Outputs (Table 4.1)**[23]. The uptake and adherence to the organic standard by supply chain stakeholders aiming to enable more sustainable agricultural production are the **Outcomes**. The actual tangible biodiversity conservation effects of implementing the organic standard are the **Impacts**.

Table 4.1
Organic standard production criteria excerpts related to biodiversity conservation.

Protecting and Restoring On-Farm Natural Habitats	Operators shall design and implement measures to maintain and improve landscape and enhance biodiversity quality, by maintaining on-farm wildlife refuge habitats or establishing them where none exist. Such habitats may include but are not limited to: a) extensive grassland such as moorlands, reed land or dryland; b) in general all areas which are not under rotation and are not heavily manured: extensive pastures, meadows, extensive grassland, extensive orchards, hedges, hedgerows, edges between agriculture and forest land, groups of trees and/or bushes, and forest and woodland.
Protecting High Conservation Value Areas	Clearing or destruction of High Conservation Value Areas is prohibited. Farming areas installed on land that has been obtained by clearing of High Conservation Value Areas in the preceding 5 years shall not be considered compliant with this standard.
Soil Erosion Prevention	Operators shall take defined and appropriate measures to prevent erosion and minimise loss of topsoil. Such measures may include, but are not limited to: minimal tillage, contour ploughing, crop selection, maintenance of soil plant cover and other management practices that conserve soil. For orchards and plantations, there shall be managed floor cover and diversity or refuge plantings.
Crop Rotation	Crop rotations for annual crops shall be established to manage pressure from pests, weeds and diseases and to maintain soil fertility, unless the operator ensures diversity in plant production by other means. Crop rotations shall be diverse and include soil-improving plants such as green manure, legumes or deep rooting plants.
Soil Fertility	Land preparation by burning vegetation or crop residues is prohibited (exceptions may be granted in cases where burning is used to suppress the spread of disease, to stimulate seed germination, to remove intractable residues, or other such exceptional cases)
Soil and Water Pollution Prevention	Stocking densities and grazing shall not degrade land or pollute water resources. This applies also to all manure management and applications.
Water Pollution Prevention	Operators shall take verifiable and effective measures to minimise the release of nutrients and waste into the aquatic ecosystem.
Water Conservation	Operators shall not deplete nor excessively exploit water resources and shall seek to preserve water quality. They shall where possible recycle rainwater and monitor water extraction.
Agrochemical Input Restrictions	Substances with high salt indexes, measured toxicity to non-target organisms, and persistent adverse effects may be prohibited or restricted in their use. Inputs used for crop production shall be considered for their impact on livestock and wildlife.
Pest, Disease and Weed Management	The organic production system relies on biological, cultural and mechanical mechanisms to manage pests, weeds, and diseases. These include protection of natural enemies of pests through provision of favourable habitat, such as hedges, nesting sites and ecological buffer zones that maintain the original vegetation to house pest predators.

Source: elaborated by the authors based on IFOAM (2014).

[23] The Organic Standard includes economic, social and environmental sustainability aspects that aim to improve producer livelihoods and incentivize more sustainable production practices.

4.3 Past performance: assessing outcome and impact

The Organic Standard assessment can be approached roughly in the same way as the **AVSS chapter**, except that it assesses a single standard instead of multiple standards with multiple criteria. Again, for this standard it is not too difficult to find data on global reach, showing the coverage and potential impact the standard can have. For level 2, quite some data is available as well, giving valuable insights in management practices. Nevertheless, coverage of these management practices is often not reported, or sometimes cannot be reported. Therefore, a level 2 division could be made based on crop types as literature insights show.

4.3.1 Level 1: Global reach

According to Willer and Lernoud (2018), land complying with in-conversion towards the organic standard is reported in 178 countries, out of which 87 have organic regulations, and occupies 57.8 million hectares of agricultural land (1.2% of the global crop land) in 2016. Three of the most important crops impacting biodiversity are wheat, maize and rice due to the extensive agricultural land dedicated to their cultivation. The total harvested wheat, maize and rice area grown under the organic standard amounted to 2.3 million ha in 2015, or 0.4% of their total harvested area (see Table 4.2) (FAOSTAT, 2017; Willer and Lernoud, 2018).

Table 4.2
Organic and conventional harvested area of wheat, maize, and rice.

		Wheat	Maize	Rice
Global Area in million ha (% of total) 2015	Conventional	220.64 (99.32%)	182.06 (99.76%)	160.52 (99.85%)
	Organic	1.52 (0.68%)	0.43 (0.24%)	0.25 (0.15%)
	Grand Total	222.16 (100.00%)	182.49 (100.00%)	160.76 (100.00%)
Compound Aggregate Growth Rate 2008–2015	Conventional	-0.07%	1.86%	0.05%
	Organic	10.23%	17.08%	27.29%
	Total	-0.02%	1.89%	0.07%

Source: elaborated by the authors based on FAO and IFOAM statistics (Food and Agriculture Organization, 2016; Willer and Lernoud, 2018).

Most of the organic staple crop cultivation occurs in Europe, where organic rice and wheat have attained 3.07% and 1.32% of the total harvested area, respectively. North America and Asia also support an important share of organic staple crop production where the standard has been most widely adopted for wheat (0.90% and 0.42% of total land for wheat cultivation respectively). It is important to mention the limited adoption of the organic standard amongst Asian rice producers (only 0.16% of total land dedicated to rice production). The greatest potential for expanding organic staple crop cultivation is found in Africa, Oceania, and Central America and the Caribbean where the organic staple crop area is below 0.01%, except for African maize which covers approximately 0.04% of cultivated area (Food and Agriculture Organization, 2016; Willer and Lernoud, 2018).

4.3.2 Level 2: Change in management practices

The Organic standard production criteria provide direct and indirect ways to conserve biodiversity by prohibiting the destruction of High Conservation Value (HCV)[24] areas, implementing measures that enhance and restore on-farm habitats, requiring soil conservation practices[25], and protecting soil and water from agrochemicals which are prohibited or limited depending on the substance. For example, the standard also restricts agrochemical use that may be harmful to biodiversity and promotes integrated pest management to enable pest predators to control potential outbreaks. Table 4.1 provides excerpts from the organic standard describing its prescriptions for biodiversity conservation, soil and water conservation and reducing agrochemical use.

Ascribing areas to these management types, however, is currently an impossible task. Sufficient information is not available on total hectares supporting specific agricultural management types such as soil conservation practices. Furthermore, these separate management requirements overlap within organic certified areas. We therefore make the assumption that these management practices equally take place over all organic certified areas, and therefore regard overall organic farming as the change in management.

4.3.3 Level 3: Biodiversity impact

The organic standard has been implemented since the 1970s enabling global meta-analysis studies to be conducted on its biodiversity impacts. Organic farming has been shown to generally increase species richness, diversity and abundance, both at the farm and landscape levels compared to conventional agriculture (Bengtsson, Ahnström, and Weibull, 2005; Katayama et al., 2019; Tuck et al., 2014). On average, organic agriculture increases on-site species richness and abundance by about 30% and 50%, respectively (Bengtsson, et al., 2005; Tuck et al., 2014). These results were highly variable in terms of the species and cropping systems examined[26]. Organic farming had the greatest effects on plant species richness, resulting in an average approximate increase of 70%, and the lowest effect on arthropod, bird and microbe species richness resulting in approximate average respective increases of 19% compared to equivalent conventional agricultural systems. In terms of cropping systems, organic farming had the greatest effect on cereal crops, resulting in an approximate average species richness increases of 42%, and the lowest effect on orchards and grass systems, whose organic cultivation resulted in approximate average species richness

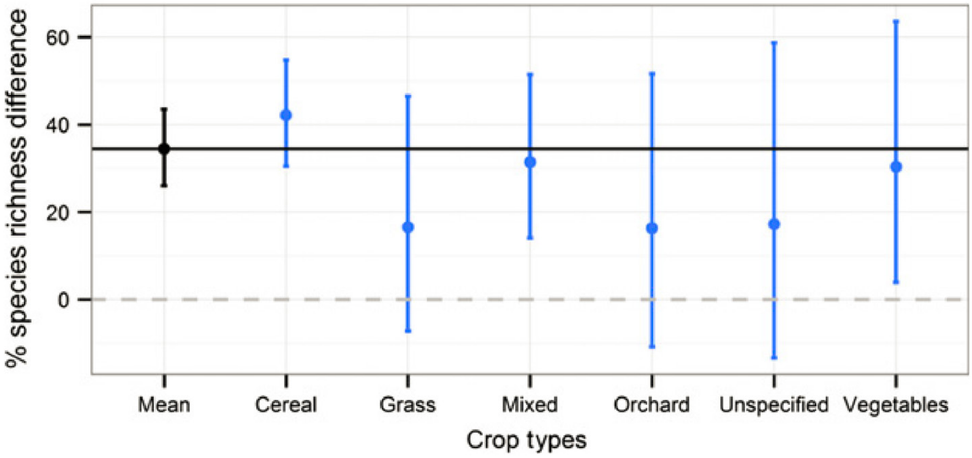
[24] Provisions to protect HCVs are not accompanied by guidance on identifying them and how compliance with this provision is audited, hence limiting the potential for the Organic standard to reduce biodiversity losses related to deforestation (Tayleur and Phalan 2016).

[25] Measures prescribed for conserving soil and water resources range from the use of minimal tillage and cover crops to prevent soil erosion, crop rotation to enhance soil fertility and lower fertiliser requirements, monitoring water extraction and recycling rainwater.

[26] The results of Bengtsson et al. were based on the meta-analysis of empirical data from 66 publications and were highly variable for species richness and abundance with 16% of the studies reviewed reporting a negative effect of organic farming on species richness. For instance, species richness and abundance generally increased for birds, insects and plants while the abundance of non-predatory insects and pests was not affected. Their findings also indicate that the landscape may have a greater effect on biodiversity than the type of agriculture practised. Organic farming had the greatest effect on species richness within intensively managed landscapes (Bengtsson, Ahnström, and Weibull 2005, 261). Due to complex interaction of local variables, the meta-analysis of empirical reviews is important to understand the overall impact of organic farming on biodiversity.

increases of 18% compared to equivalent conventional agricultural systems (Katayama et al., 2019; Tuck et al., 2014) (see Figure 4.2).

Figure 4.2
The difference (increase) in species richness (%) on organic farms relative to conventional farms, classified by crop types.



The grand mean is shown in black, accompanied by the black horizontal line. The dashed line indicated the zero line (no difference). The blue lines indicate 95% credible intervals, which were calculated from posterior standard errors. (Source: Tuck et al., 2014, p. 51).

Improved biodiversity associated with organic agriculture can be partly attributed to lower fertiliser and pesticide applications, estimated to be a respective 34% and 97% lower, compared to conventional agricultural production systems (Maeder et al., 2002). Reliance on organic fertilisers has been related to the higher abundance and performance of both pests and pest predators resulting in more arthropod biodiversity and crop system resilience to pest attacks (Garratt et al., 2011). Smith et al., (2011) support that production practices that tend towards diversity and away from intensification are better for supporting biodiversity and that these benefits often extend beyond farm boundaries as it enables 'self-sufficient ecosystems' within the farm to lessen dependence on surrounding natural habitats to sustain species pools and production[27]. For this reason, the benefits of organic farming are often more pronounced in intensively farmed landscapes (Bengtsson, Ahnström, and Weibull, 2005; Tuck et al., 2014).

Organic wheat is examined more closely to determine its biodiversity conservation potential. Organically grown cereal crops, including wheat, on average increase species richness by 42% compared to conventionally grown cereal crops indicating that the organic standard may have greater potential for conserving biodiversity within wheat cultivation systems (see Figure 4.2) (Tuck et al., 2014).

The studies summarised above convey that farming compliant with the organic standard improves

[27] Smith et. Al., (2011) reviewed 82 studies at plot, field and farm scales to find that organic farming significantly benefits biodiversity, especially of plants. These authors reported as particularly beneficial: avoidance of agrochemicals, crop rotation, mixed cropping and green manure, maintenance of on-farm habitats (through permanent pastures and hedgerows), rational application of slurry and manure, and mixed livestock-crop enterprises. These results were, however, highly dependent on a diversity of uncontrolled variables.

species richness and abundance depending on the species and cropping systems examined. Mean species abundance estimates could be derived from these studies to establish the biodiversity benefits of organic agriculture. Nonetheless, the yield gap between organic and conventional farming needs to be taken into consideration to adequately assess the biodiversity conservation potential of the organic standard (see Section 4.4 Discussion and Conclusion). Conclusive evidence on the impact of organic agriculture on biodiversity exists in the form of long-term meta-analysis studies, which have been conducted for the most part in developed countries located in more temperate climates but studies are lacking in developing countries located in tropical climates.

4.3.4 Level 4: Implementation effectiveness

Unlike implementation effectiveness challenges encountered by the previous chapter in AVSS, this chapter on Organic Agriculture Certification has fewer barriers to overcome as far as Level 4 is concerned. Despite this, as the 'Discussion and Conclusion' section points out, the challenges are adequate enough to preclude a concise assessment of implementation effectiveness of organic agriculture certification. In the case of organic agriculture certification, there are three major reasons due to which an undertaking Level 4 is not conducive.

Firstly, due to the indirect contribution of organic agriculture certification towards biodiversity conservation it is difficult to gauge the implementation effectiveness of organic agriculture accordingly. This chapter also states that even though organic certification has far reaching effects on carbon sequestration and biodiversity, the distinction between the two is not clear.

Secondly, this chapter argues that the actual coverage area under organic agriculture certification has increased at the time of this study. The ever changing nature of areas brought under organic agriculture certification makes it difficult to gauge the implementation effectiveness. However, this study does identify organic agriculture certification as one of the crucial ICIBs alongside others for biodiversity conservation.

Thirdly, this chapter states that there is far too limited data — due to lack of research — on the impacts of organic agriculture certification in developing and under-developed countries. As most data on organic certification currently exists in developed countries, a holistic impact assessment cannot be conclusively carried out. This makes it particularly challenging to understand the implementation effectiveness of organic agriculture certification on a global scale. These details are further elaborated in the following section.

4.4 Discussion and conclusion

Organic certification occupied 57.8 million hectares of agricultural land (1.2% of the global crop land) in 2016. The total harvested wheat, maize, and rice area grown under the organic standard amounted to 2.3 million ha in 2015, or 0.4% of the total harvested area of these crops. Organically grown staple crops have been expanding much faster than conventionally grown staple crops based on examining harvested areas from 2008 to 2015. Organic compliant harvested area experienced double digits CAGR compared to a slight increase or decline in CAGR for the conventional harvested area staple crops[28]. However, organic compliant harvested area still represents at best 1% of the total land allocated to the staple crops, even though the organic standard is fairly ubiquitous, operating only in 25% to 50% of the staple growing countries across the world[29]. This points to an important opportunity to expand the overall presence of the organic standard in staple crops. With provisions for conserving natural habitats, soil and water resources and reducing agrochemical use, and the fact that 43% of all agricultural land is dedicated to the cultivation of these crops (wheat is the most widely distributed food crop with 220 million ha, followed by maize, with 195 million ha, and rice with 165 million ha) (Food and Agriculture Organization, 2016; Tayleur et al., 2016)[30]. The organic standard is well-positioned to expand and conserve biodiversity in the staple crops. The biodiversity conservation potential within these crops depends on the land converted and agricultural practices adopted for their cultivation:

- Wheat has been one of the 'most important sources of biodiversity loss caused by agriculture' as vast grasslands, now threatened from extinction, have been lost to give way to its cultivation (Potts et al., 2016; Gibson, 2009). Wild and domesticated wheat varieties, increasingly replaced by commercial ones, are further threatened by the introduction of genetically modified wheat. This development may limit organic wheat expansion as the standard prohibits genetically modified organisms (GMOs) (Heinemann, Agapito-Tenfen, and Carman, 2013).
- The adaptability of maize has led to its cultivation in diverse environments which typically leads to natural habitat losses, soil depletion and erosion, and fertiliser runoff which harms aquatic biodiversity. The proliferation of GMO maize is also a concern for preserving wild and domesticated varieties and expanding organic maize. Climate change offers an opportunity for organic maize to expand as it may be better adapted to harsher conditions particularly in tropical climates (McKie, 2017; Pimentel and Burgess, 2014).
- Rice fields represent 15% of the world's wetlands and support water birds (Ellis, 2014). However, rice cultivation has resulted in natural wetland losses and degradation, soil pollution and water depletion as it appropriates 34% to 43% global irrigation water. After deforestation, wet rice cultivation is the largest greenhouse gas emitter in South and Southeast Asia (Ellis, 2014; Kritee et al., 2018; van Groenigen, van Kessel, and Hungate, 2013).

While no final calculation could be made on the biodiversity impact of organic certification due to a lack of MSA categories for organic certification, there is significant field-level evidence that shows that organic farming conserves biodiversity. The meta-analysis reviewed generally reports an average 30% increase in species richness and 50% increase in species abundance for organic agriculture compared to conventional agriculture (Tuck et al., 2014). Biodiversity conservation benefits will depend on the contexts in which organic agriculture is practised. For instance, organic

[28] The cumulative aggregated growth rate for Organic compliant harvested area from 2008 to 2015 was the following: wheat = 10%, maize = 17% and rice = 27%. The cumulative aggregated growth rate for conventional harvested area from 2008 to 2015 was the following: wheat = -0.1%, maize = -2%, rice = 0.05.

[29] The total area dedicated to wheat, maize and rice cultivation in 2015 was 222, 183 and 161 million hectares respectively out of which 1.5, 0.4 and 0.2 million hectares respectively were Organic compliant. Organic production can be found in 59 of the 123 wheat growing countries, 42 of the 168 maize growing countries and 33 of the 118 rice growing countries.

[30] Wheat, corn and rice occupy 13%, 11% and 10% respectively of the world's agricultural land (Food and Agriculture Organization 2016).

agriculture practised in the cereal crops results in an average increase in species richness of 42% while in orchards and vineyards it results in an average species richness improvement of 18% (Katayama et al., 2019; Tuck et al., 2014). The meta-analyses reviewed warn that studies from Asia, Africa, and Latin America are lacking for both biodiversity and yield gap assessments. If sufficient evidence becomes available on the impact of organic certification on biodiversity, these numbers could be used to create MSA categories which would be significantly beneficial in computing MSA values for organic certification.

Despite clear indications that organic agriculture benefits biodiversity, its expansion in the staple crops has been limited due to its lower yields. Meta-analysis studies on yield differences report a drop of 19% to 26% for organic agriculture compared to conventional agriculture attributed to agronomic reasons such as plant fertilisation differences and maladapted planting material and experimental reasons including prevalence of data from developed countries and impact assessment research designs that disregard the multi-crop and rotational complexities of organic farming (Seufert, 2019). Expanding organic farming in the staple crops to conserve biodiversity will require overcoming the yield gap with conventional agriculture. Enhancing fertilisation, improving nutrient availability, better weed management practices and crop variety development (up to 95% of the cultivars used in organic farming were bred for conventional farming systems) could go a long way towards addressing the organic and conventional farming yield gap (Knapp and Heijden, 2018; Luo et al., 2018; Seufert, 2019). It must be noted that most of the yield gap research has been conducted in developed countries, where conventional agriculture is particularly input-intensive[31]. Furthermore, a more holistic yield assessment taking into consideration multi-cropping system yields (e.g. agroforestry) and crop rotations would result in higher yields for organic farming and further close the yield gap (Seufert, 2019)[32].

Although the environmental impacts of organic agriculture may be lower per unit area, its inferior yields implies that its environmental impacts may be higher per unit crop (i.e. nitrogen-related pollution, land use, eutrophication and acidification) (Tuomisto et al., 2012). Clarke and Tilman, (2017) report that organic farming uses less energy than conventional agriculture but emits a similar amount of greenhouse gases, requires more land and causes more eutrophication per unit crop[33]. They conclude that adopting low-impact plant-based foods as opposed to meat and increasing agricultural input efficiency could result in more environmental benefits than a global shift from conventional to organic farming. However, this climate mitigation focused assessment disregards the intrinsic value of biodiversity, the ecosystem services that it supports and its potential role in climate change adaptation. For instance, the ecosystem service values provided by organic versus conventional agricultural landscapes in New Zealand were estimated at USD 1,610 to 19,420 ha/yr

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[31] Since a significant portion of wheat is grown in Europe and North America applying the general findings reported above on the potential for organic farming to conserve biodiversity is further warranted.

[32] This assessment is not feasible due to conventional farming reliance on monoculture and limited rotation (Seufert, 2019).

[33] Similar results were reported for nitrous oxide emissions by Skinner et al. (2014), who claimed that to equalise the mean "difference in yield-scaled nitrous oxide emissions between both farming systems, the yield gap has to be less than 17%" (Skinner et al. 2014, 468). Others, like El-Hage Scialabba and Muller-Lindenlauf (2010) disregard the lower yield issue to estimate that converting all the world's agricultural production to organic would reduce its GHG emissions by 20% in terms of avoided fertiliser-related energy consumption and NO3 emissions. They also estimate that organic croplands and grasslands could sequester carbon equivalent to 40% to 72% of the 2010 global agricultural GHG emissions.

and USD 460 to 14,570 ha/yr, respectively (Sandhu et. al., 2008)[34]. Organic farming in China resulted in a saving on agricultural inputs of USD 518.3 million (or USD 447.7 per ha) and ecosystem services worth USD 320.2 million (or USD 276.5 per ha) — with USD 47.8 million from increased biodiversity and the rest from carbon sequestration and nitrate leaching reductions. These input savings and ecosystem services (USD 1,019.2 million, or USD 880 per ha) almost close the yield gap between organic and conventional farming which was assumed to be 10% to 15% highlighting the importance for valuing the additional benefits of organic agriculture (Meng et al., 2017). Nonetheless, the organic standard is well positioned to enable more biodiversity friendly staple crop cultivation due to its strong market growth, ubiquitous global presence and biodiversity supporting production criteria.

[34] The ecosystem services valued by Sandhu et al. (2008) included pest control, soil formation, mineralisation of plant nutrients, pollination, aesthetics, food and material provision and carbon accumulation.

5 Community Forest Management (CFM)

Community Forest Management (CFM) has become an influential approach in the management of forests by Indigenous People and Local Communities (IPLC) around the world in the last couple of decades (Agrawal, 2001; Arnold, 2001; Di Girolami et al., 2023; Hajjar et al., 2021; Wiersum, 2009). Nearly 15% of global forests — about 500 million hectares — fell under the Community Forest Management arrangement in 2013 (Rights and Resources Initiative, 2014). As a response to state forestry and commercial timber production, and building upon traditions of customary regulation of forest commons, this approach puts fulfilment of local livelihoods and forest conservation first. In general, CFM can be defined as the use, management and conservation of forests by indigenous communities. Such forests may, may not, or may be partially owned by communities, and their management is often practised in various degrees of collaboration with state forest agencies, donor organisations, knowledge institutions and/or companies. Because of this variation, several terminologies are used to refer to these practices (community forestry, community-based forestry, community-managed forests, community forest management, collaborative forest management, participatory forest management, joint forest management and forest co-management). Here we use the term 'Community Forest Management', or CFM, because it is most referred to in the literature after the term 'community forestry' (based on a Google Scholar search) and since we prefer to avoid using the word 'forestry' in this context, because its connotation is largely related to timber production, and not so much to forest management in a broader sense, including biodiversity conservation.

The central idea behind CFM is that local management of forests, either by communities or jointly with forest departments, is more effective in providing sustainable use and local livelihoods than management by central state institutions. CFM brings a new 'sense of ownership', either legal or practical, and hence, new responsibilities and dignities to people. This particularly applies to developing countries in the tropics, where state institutions are often weak in forested landscapes, or even absent in remote areas. While restricted access to and sustainable management of state forests are regulated on paper, open-access regimes and tragedies of the commons remain in practice (Hardin, 1969). In such cases, CFM can bring an attractive alternative for state forestry and open-access regimes.

Already in the early 1970s, the idea of community participation, both for better forest management and for improving people's livelihoods, was practised in a few countries, advocated by NGOs and scientists, and intensively discussed within the Food and Agricultural Organization (FAO) at global level (Arnold, 2001; Food and Agricultural Organization, 1978; Umans, 1993). Later, these ideas became norms within international law, both as hard and soft law, e.g. in Agenda 21, the Rio Forest Principles, the Convention on Biological Diversity and the Non-Legally Binding Instrument on All Types of Forests (Arts and Babili, 2013). Such ideas and norms have in turn travelled to national levels, where they became embedded in forest law and policy, or strengthened already existing local CFM practices in countries. For example, India, Nepal, Mexico, Bolivia, Kenya, and Tanzania have pioneered different forms of CFM from the 1980s onwards and many countries, from Ethiopia to Albania, followed later (Baynes et al., 2015; Charnley and Poe, 2007).

The history of CFM exhibits various phases in which different approaches were experimented with. Wiersum (2009) distinguishes the following between 1970 and 2010: (1) a conservation and basic needs phase, in which CFM mainly targeted the conservation and rehabilitation of community forests, and basic needs; (2) an empowerment phase, in which the democratic and forest rights of local communities were emphasised; (3) a collaborative phase, in which cooperation and joint decision-making of state agencies, donors and local communities were put central stage in order to alleviate poverty and sustainably manage forests; and (4) an entrepreneurial phase, in which CFM initiatives have been related to the establishment of local enterprises and to global value chains, including community forest certification (Wiersum et al., 2013). These phases were not perfect corollaries of each other, and did not neatly follow each other. Instead, they overlapped with each other due to which many aspects of these various phases of CFM still exist in parallel today.

5.1 Motives, goals, targets

The overall **motive** for CFM is to prevent deforestation and forest degradation, while improving forest-dependent livelihoods at the same time (Arts and De Koning, 2017). As such, it contrasts the classical forest conservation narrative of protected areas without people ('fortress nature') (Palomo et al., 2014). The latter has often led to exclusion of people from their lands and violation of their forest rights, thus fuelling debates on 'doing conservation otherwise' (Dressler et al., 2010). Consequently, discourses on proper forest management and conservation drastically shifted over time (Umans, 1993; Wiersum, 2009).

Overall, CFM exhibits two **goals**: (1) To enhance the sustainable management of community forests; and (2) To improve forest-related livelihoods for IPLC. Sometimes, a third goal is also strived for: (3) To empower IPLC vis-a-vis the state. In this case study, we focus on the first two.

lthough CFM does not adhere to specific global **targets**, it contributes to several. The ones that come closest are some of the Kunming-Montreal targets of the CBD, particularly Target 14 (full integration of biodiversity values in development and poverty reduction strategies), Target 1 (ensure that all areas are under participatory integrated biodiversity inclusive spatial planning and/or effective management processes addressing land and sea use change, to bring the loss of areas of high biodiversity importance, including ecosystems of high ecological integrity, close to zero by 2030), Target 10 (ensure that areas under agriculture, aquaculture, fisheries and forestry are managed sustainably, in particular through the sustainable use of biodiversity), Target 11 (restore, maintain and enhance nature's contributions to people, including ecosystem functions and services, such as regulation of air, water, and climate, soil health, pollination and reduction of disease risk) and Target 21 (traditional knowledge of local and indigenous communities are respected and integrated in national and international biodiversity policies). CFM is also linked to Sustainable Development Goal (SDG) 15 (protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, halt and reverse land degradation and halt biodiversity loss).

5.2 Theory of change: from input to impact

CFM as a practice resulted from: (1) 'changes in forest ownership', from the state to IPLC, followed by proper forest management planning (Ostrom et al., 2002), or (2) 'awareness raising' of the need of local sustainable forest management, followed by an agreement of mutual rights and responsibilities between authorities and IPLC in community or state-owned forests (Arts and Babili, 2013). Subsequently, the theory of change assumes that forests, if owned or managed by local people themselves, will lead to more responsible behaviour for sustaining the resource and to higher stakes in conserving it. Thus, both ownership and/or management rights will ideally lead to sustainable forest management practices, while substantially reducing or even halting deforestation and forest degradation. The results will be improved livelihoods and improved forest conditions, the latter including positive biodiversity impacts as well.

5.3 Past performance: assessing outcome and impact

5.3.1 Level 1: Global reach of CFM

For level 1 analysis, we work with three data sets: the 2015 Food and Agricultural Organization (FAO) Global Forest Resources Assessment, which — contrary to more recent Global Forest Resources Assessments — includes data on forests managed and/or owned by IPLC (Food and Agricultural Organization, 2015), the Rights and Resources Initiative (RRI) on forest tenure, including data on IPLC (Rights and Resources Initiative, 2014) and FAO's 2016 review of 40 years of community-based forestry (Food and Agricultural Organization, 2016b). We decided to work with these three data sets, and not just one, in order to increase the robustness of evidence, as there are plenty of measurement and data uncertainties at the global aggregate level. Moreover, several conceptualisations of CFM are used in these three data sets. Now, by combining those, we construct a *hypothetical trend of CFM area over time*, including margins of uncertainty. However, this approach exhibits some downsides. First, these databases do partially overlap. RRI 2014 includes FAO data from 2010 in its analysis, as does FAO 2015 with the addition of national information sources and independent expert sources. In addition to this, FAO 2016 includes updated data from FAO 2010 and RRI 2014 as well, amongst other data sets (e.g. a database from RECOFTC). At the same time — given their differences in definitions, data sets and data processing — they might neither be fully compatible. But we deem the pros of the approach (more robust evidence, addressing uncertainty) more important than its cons (overlaps and differences). Over time, we checked and updated Tables 5.2 and 5.3 thrice, in order to avoid errors and improve the approach (October 2016, April 2019, November 2022). This also explains the differences with our earlier assessment (see Arts et al., 2017).

Table 5.1
Overview of rough data and omissions

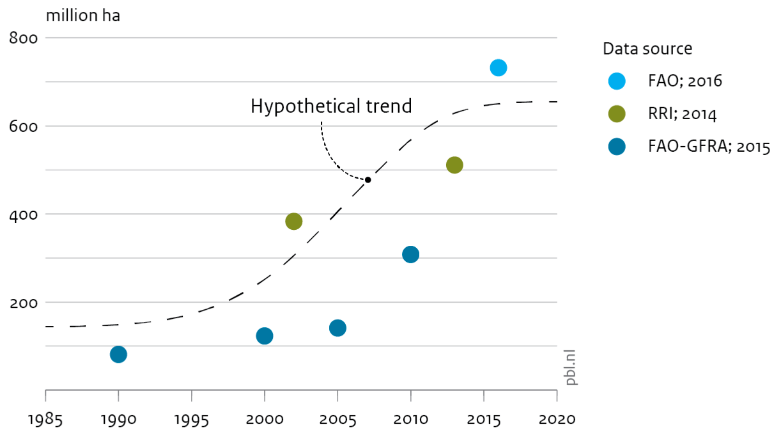
Global forest area owned and/ or managed by IPLC (million hectares)			
	FAO-GFRA, 2015	**RRI, 2014**	**FAO, 2016**
1990	81	-	-
2000	123	-	-
2002	-	383	-
2005	141		-
2010	308	-	-
2013	-	511	-
2016	-	-	732

From Table 5.1, we learn that – according to FAO data – IPCL owned and/or managed about 100 million ha of forests in the 1990s (which amounts to about 2.5% of global forest cover), about 300 million ha. in 2010 (7.58% of global forest cover), and more than 700 million hectares in 2016 (17.5% of global forest cover). This is a steep curve over time, but as said, definitions and data sets changed over time in FAO reports. RRI data shows another upward trend, from nearly 400 million ha. of CFM forest area in 2002 to about 500 million ha. in 2013 (which amounts to about 12.5% of global forest cover). Below, in Figure 5.1, we will try to make sense of these different sources, data sets, trends and uncertainties, but first we need to say a few words about what these figures mean for CFM in relation to potential biodiversity impact.

For the time being, based on Table 5.1, while simply combining the three data sets without taking all limitations and uncertainties into account yet, **we assume that the total CFM area worldwide amounted to about 500 to 700 million ha. in 2015**. However, not all of these forests are probably used and managed based on the latest sustainable forest management principles and insights, including biodiversity conservation. After all, many 'forest commons' are a heritage of the past, while others have been recently installed or re-introduced to improve forest management. We assume that the 'old' forest commons are still used or managed according to customary or business-as-usual practices. This may or may not include biodiversity considerations. Since including such considerations explicitly emerged in the 1980s only (Arnold, 2001; Food and Agricultural Organization, 2016b; Wiersum, 2009), we decided to exclude the pre-1980 'old' forest commons from our estimates below.

Figure 5.1

Global forest area owned or managed by Indigenous People and Local Communities



Annual expansion of forest owned and/or managed by indigenous people and local communities per data set and per decade.

However, 1980s data are not available in Table 5.1, but would be interesting to have, since these will deliver the insights to exclude 'CFM without biodiversity' in our estimates. To retrieve those, we decided to plot the data of Table 5.1 in Figure 5.1, and to deduce and draw a hypothetical longitudinal trend of global CFM area over time, ranging from the 1980s until today, while the different data dots around the curve indicate some margins of uncertainty. This trend is definitely hypothetical, because of the different conceptualisations and limitations behind these three data sets. At first sight, the dots in the plot seem to refer to an exponential curve over time, but insights from the literature made us decide to draw an S-curve. After all, several sources indicate that CFM has shown initial modest growth in its early stages, then a steep expansion curve for quite some time, and more recently, a more modest growth again (Arnold, 2001; Rights and Resources Initiative, 2014; Wiersum, 2009). Also, we decided to draw the line closer to the RRI data than to the ones of FAO, because the former are generally considered more robust than the latter (IFRI, 2015). Given these hypothetical curves and margins of uncertainty, we assume that the CFM area increased from about 100 to 200 million ha. in the 1980s to about 600 to 700 million ha. today. **Such implies that we can estimate that the post-1980 CFM area — thus including biodiversity conservation goals — amounts to about 400 to 600 million ha. today**.

5.3.2 Level 2: Change in management practices under CFM

To assess the biodiversity impact of CFM, we need to know the change in management practices that take place under CFM, and the increase of forest biodiversity as a consequence of this shift. We distinguish three possible trajectories of such change on the basis of academic literature, supervision of field studies, and field observations in various countries (see Arnold, 2001; Arts and De Koning, 2017; Charnley and Poe, 2007; Di Girolami, 2019; Food and Agricultural Organization, 2016b; IFRI, 2015; Wiersum, 2009). A first trajectory is that communities **restore degraded forests** by planting new trees. This is quite a common practice in CFM. A second trajectory is that communities **enrich poor forests or plantations** towards a more diverse, secondary forest by fostering natural regeneration, by enrichment planting and by careful planning and management of forest use. Such management practices not only enhances forest biodiversity, but also advances new forms of sustainable use, including possibly timber production. And a third trajectory concerns the **shift from secondary to man-made, semi-natural forests, again by similar kinds of forest management techniques** (e.g. enhance natural regeneration, enrichment planting). Although 'semi-natural', this still includes forest use and, possibly, timber production. Therefore, we exclude the potential shift to man-made 'natural' forests, because we look at sustainable use under CFM, including the delivery of timber, NTFPs and biodiversity, not at the sole protection of natural forests by communities (which is called 'community-based conservation', or CC, yet another category of community involvement than CFM; see Chapter 10 in this report).

However, not all CFM aims at forest enrichment for local development and conservation; a large proportion also tries to maintain the current situation, hence tries **to avoid further deforestation and forest degradation** in a broader landscape context where forests are disappearing and/or deteriorating (the latter is also abbreviated as ADD, 'avoided deforestation and degradation'). ADD is probably as common in CFM practices as forest enrichment. In worst cases, CFM might even create a 'forest island' in a deforested landscape, where CFM has prevented the conversion of forests to other land uses. Considering ADD in the context of CFM, various types of forests can be thought of as being prevented from deforestation or further degradation: plantations, secondary forests, and semi-natural forests (see the categories in the above). Again, we exclude 'natural forests' here, because we look at CFM, not CC. **All in all, we identify six management practices of forest change and ADD under CFM**: the three 'change trajectories' as shown in the above (degraded forest --> plantation; plantation --> secondary forest; secondary forest --> semi-natural forest) and 'three maintenance' trajectories of ADD (maintenance of plantations, secondary forests and semi-natural forests, respectively). **Ideally, we would like to know area coverage of each of these management practices and trajectories — as proportions of the total global CFM area —, but these data are, for as far as we know, not available in the literature, unfortunately**.

5.3.3 Level 3: Biodiversity impacts under CFM

To calculate the actual biodiversity impact of CFM, we connect the different management practices and forest change trajectories from the above to the MSA indicator (see Chapters 1 and 2 of this report). In Schipper et al., (2016), a natural forest shows a biodiversity score of 1.0 and a lightly used forest is set at 0.70 (see Table 5.2). This means that about 70% of the species populations of flora and fauna that are usually present in an undisturbed natural forest can still be found in lightly used or managed forests. Table 5.2 shows the MSA level of more and less managed and disturbed forests.

Table 5.2
MSA values assigned to GLOBIO land-use classes.

GLOBIO land-use class	MSALU
Forest – Natural	1.0
Forest – Plantation	0.30
Forest – Clear-cut harvesting	0.50
Forest – Selective logging	0.70
Forest – Reduced impact logging	0.85

Sources: Alkemade et al., 2009; Alkemade et al., 2013; GLOBIO reference database (www.globio.info)

In a next step, we link the management practices and forest change trajectories identified in the previous section to the MSA forest values of the GLOBIO land-use class model (see Table 5.3). **On average, MSA increases with 0.2 for forest enrichment under CFM, and MSA is maintained at 0.5 MSA on average for CFM as ADD**. Of course, such a 20 percentage points increase in forest biodiversity is not immediately realised once a CFM arrangement of forest enrichment is installed. Based on knowledge of various cases, this might take between 10 to 30 years to materialise (depending on the type of management practice, type of forest change trajectory and the type of forest). By contrast, maintenance of MSA under CFM/ADD can be immediately realised, once an initiative starts.

Table 5.3
MSA gains through various CFM trajectories

	State of forests before CFM	MSA	State of forests after CFM	MSA	MSA Effect
Trajectory 1	Degraded, exploited	0.1	Plantation forestry	0.3	+0.2
Trajectory 2	Plantation forestry	0.3	Managed, secondary forest	0.5	+0.2
Trajectory 3	Managed, secondary forest	0.5	Managed, semi-natural forest	0.7	+0.2
Trajectory 4	Plantation	0.3	Plantation 'avoided loss'	0.3	+0.3
Trajectory 5	Secondary forest	0.5	Secondary forest 'avoided loss'	0.5	+0.5
Trajectory 6	Semi-natural forest	0.7	Semi-natural forest 'avoided loss'	0.7	+0.7

Note that we left out 'RIL in near-natural forests' (MSA +0.85) as an option for CFM, besides natural forests. RIL or 'Reduced Impact Logging' is a high-tech forest management approach to minimise forest damage of selective logging (Putz et al., 2008). It for example includes techniques like GIS mapping, digital tree identification at individual level and smart harvesting. Although some modes of CFM might include such high-tech operations, like Community Forestry Enterprises (CFEs) in countries like Bolivia, Brazil, and Cameroon, this is probably not common practice in CFM yet. To conclude this part, **CFM includes three management trajectories of enhancing forest biodiversity, with an increase of 0.2 MSA on average, and three conservation trajectories of ADD, with 0.5 MSA conservation on average**. However, since we do not know the forest area coverage of these various practices, we are unable to estimate the average Ha/MSA gain through CFM in general.

5.3.4 Level 4: Implementation effectiveness of CFM

Of course, not all forest management practices under the 'new' CFM regime do actually contribute to sustainable forest management and biodiversity conservation on the ground, as numerous field studies from all over the world show that the results of CFM policies, programmes and projects on the ground are generally mixed (Arts and De Koning, 2017; Charnley and Poe, 2007). In other words, 'implementation gaps' do occur. To gain some insights about the effectiveness of CFM, we executed a systematic literature review (Di Girolami, 2019; Di Girolami et al., 2023). This review identified over 2,000 potential titles, but — after thorough screening — only included 36 robust impact studies. Through these, 89 assessments on flora, fauna and ecosystem services, representing about 6 million ha., or about 1% of CFM forests around the world, were reviewed and synthesised. Indicators for environmental impact in these studies refer for example to basal area, canopy cover, deforestation, forest biodiversity, forest degradation, forest growth, forest regeneration, wildlife, biomass, carbon stock, and others. Now 72 of those assessments report positive environmental impacts on one or more of these indicators, 14 show no impact, whereas 3 assessments report a negative impact. These figures imply an 'implementation effectiveness' of about 80% (72 out of 89), which seems quite a high performance for PFM. To test for 'positive bias', only those studies with the highest quality and rigour values were taken out (Di Girolami et al., 2023). On doing so, the effectiveness drops a bit (about 70%; 10 out of 14 assessments report positive impacts). Consequently, **it can be concluded that about three quarters of available environmental impact assessments on CFM report more or less positive results**. Although, one should be careful in generalising these findings to all CFM forests, given the 1% coverage of these impact studies.

5.4 Discussion and conclusion

The findings of this chapter are as follows:

- **Global reach of CFM** that includes biodiversity goals is estimated to amount to about **400 to 600 million ha.** today (through sustainable forest management and use by IPLC).
- CFM includes various changes in management practices: three trajectories of enhancing forest biodiversity, with an estimated increase of **0.2 MSA on average**, and three trajectories of ADD ('avoided deforestation and degradation'), with an average **0.5 MSA gain** being estimated.
- **Implementation effectiveness** is — for as far as impact studies are available — reported to be about **70% to 80%** (in other words, about three quarters of impact assessments of CFM report more or less positive biodiversity impact on the ground).

However, all these figures show substantial bandwidths, come with many uncertainties and data gaps, do not fully represent all CFM forests around the world, and are based on various assumptions (on CFM expansion over time, on the nature of different management practices and, on implementation effectiveness gaps). This implies that these figures should be considered **indicative and preliminary** at best.

But more importantly than producing exact figures, this chapter was meant to experiment with the 'ICIB method' as developed in Chapter 2. In that respect, this chapter fulfils its aim rather successfully. We could work well with the method, and it therefore offers a promising tool for assessing the biodiversity impact of ICIBs (see Chapter 11 for a more thorough discussion of the approach).

6 Forest Certification

The demand for timber products is a major cause of tropical deforestation, forest degradation and biodiversity loss. These effects occur directly through forest exploitation and conversion, and indirectly through human disturbances such as habitat fragmentation, infrastructure, pollution, greenhouse gas emissions and so on. Deforestation and forest degradation reduce the capability of forests to deliver important ecosystem services such as carbon sequestration, local climate regulation, soil retention, firewood provision and water regulation (Fisher et al., 2009). Sustainable Forest Management (SFM) can reduce the loss of forest biodiversity and can add to maintaining benefits to local livelihoods. An important governance mechanism to stimulate SFM is using Voluntary Sustainability Standards (VSS) for forest management.

Voluntary Sustainability Standards were originally initiated by civil society NGOs in industrialised countries, concerned with environmental and social development, especially to provide a non-governmental governance arrangement for international cross-boundary trade. Often in collaboration with market parties, they aimed to raise awareness about sourcing sustainable products amongst conscious consumers. This is in line with the Kunming-Montreal Target 10: Ensuring that areas under agriculture, aquaculture, fisheries and forestry are managed sustainably, including through a substantial increase of the application of biodiversity friendly practices. By setting standards for improved production methods, working with local producers and by introducing product labels to influence consumer choice, the path was set to increase sustainable production and consumption (Vermeulen and Kok, 2012).

One of the first initiatives was the creation of Voluntary Sustainability Standards for forest commodities. During the 'tropical timber crisis' in the 1980s, environmental organisations put the spotlight on the continuous and critical deforestation process in tropical countries. One important cause for this was the consumption of high-quality tropical timber by developed countries. This realisation resulted in environmental groups calling for the boycott of tropical timber; however, this was later seen as a counterproductive strategy (Meidinger, 2006). As a replacement for the tropical timber boycott, ideas about a system for certifying sustainable forest management were put forward. Such a system would enable consumers to purchase wood from sustainably managed sources, thereby avoiding illegal deforestation and unsustainable forestry (Meidinger, 2006). After the failure at the Rio Earth Summit (1992) to achieve a highly expected binding forest convention and as a response to the lack of national and international legislation and coordination on environmental and human rights, a large group of NGOs, led by the Worldwide Fund for Nature, founded the Forest Stewardship Council (FSC), also reflecting the increasing movement from government to governance (Overdevest, 2010).

Currently FSC is a global certification scheme managed by a non-profit organisation that creates standards for socially and environmentally sustainable managed forests through consulting with their stakeholder network and its member base holding forest concessions. By the end of 2019, FSC certified forests covered about 200 million hectares (FSC, 2019), representing 5% of the 4.06 billion hectares of forests worldwide in 2020 (Food and Agricultural Organization, 2020). The international standard contains general principles that can be applied to any forest throughout the world, and FSC acknowledges and approves more specific national and regional standards that adapt the general principles to the local conditions (Overdevest, 2010). The 10 FSC principles cover subjects like compliance with national and international laws, the social and economic rights of workers and communities, rights of indigenous people and local communities to use the forest, plantation establishment, environmental impact and identification of High Conservation Values for set-aside strategies and so on[35]. Third-party auditing of local implementation of the applicable standard at wood production sites is required before products can be produced, traded and sold with the FSC trademark.

Another, comparable initiative, the Programme for the Endorsement of Forest Certification (PEFC) was launched in 1999 by representatives of national PEFC governing bodies, mainly formed by landowners and the forestry industry (Vermeulen and Kok, 2012). It was conceived as a European umbrella organisation with a mandate to evaluate and endorse national standards for sustainable forest management. So instead of developing one general standard for all forests, they endorse standards developed at various national levels of respective nations, in compliance with PEFC International's sustainability benchmark. The PEFC benchmark covers, among others, the conservation of biodiversity, worker and community rights, local employment, plantation establishment and compliance with the law (https://pefc.org/standards-implementation/standards-and-guides). The establishment of PEFC was initially regarded as a counter reaction to FSC, but the two systems are ever more converging over time. When FSC national standards were established in countries like Finland and Sweden in the 1990s, there was strong resistance from small, farm-forestry operators concerned with the protection of private property rights and with minimising costs for small-scale forestry operators (Cashore et al., 2005). Originally it was an European market-led initiative, but by now, its governance structure is made-up of national members as well as international stakeholders such as NGOs, businesses and associations at the global level (www.pefc.org). The first national schemes were endorsed by the PEFC in 2000. Today over 300 million hectares of forests are PEFC certified worldwide.

For this case study on the biodiversity effects of forestry certification we will look at the impacts of both FSC and PEFC as the two largest ICIBs in the forestry sector.

6.1 Motives, goals and targets

The **motive** of both FSC and PEFC is to maintain and use forest resources worldwide in a sustainable way by supplying guidance to forest owners and concessionaires to manage the natural forest resources in such a way that the different functions are not compromised.

The **goal** of forest management certification is to differentiate between timber products produced in accordance with social and environmentally sustainable standards and timber products that are produced conventionally without the control of such standards. The standards codify the practice of sustainable forestry in ways that support the widely held views and agreements of societal stakeholders on reducing the decline in forest resources and supporting local livelihoods, while at the same time converting consumer demand for more sustainable products into forest management-level incentives (Milder et al., 2015).

Next to certifying the forest management area, the supply chain itself also has to be certified, to guarantee the traceability of final wood products to the certified sourcing areas. The FSC and PEFC

[35] https://ic.fsc.org/en/what-is-fsc-certification/principles-criteria/fscs-10-principles

standards use additional criteria and control procedures to make this sure, which is a requirement for wood entering the EU market (acc. FLEGT regulation). This type of certification will not be treated here further, as it is not directly related to biodiversity.

The **targets of FSC and PEFC are similar but with different nuances**. In general, VSSs aim to implement internationally agreed upon principles, into norms and criteria which can be measured, monitored and verified with indicators at concession level (SCSKASC, 2012). Both FSC and PEFC use their own definitions of sustainability in their standards, which results in a variation in principles, criteria and indicators that are used to certify conformity (PCI; van Bueren and Blom, 1997). Two important Kunming-Montreal GBF targets that are relevant for forest certification are target 5 'Rate of forest loss and rate of habitat loss halved in 2020 and, if feasible, close to zero' and target 10 'Ensure that areas under agriculture, aquaculture, fisheries and forestry are managed sustainably'. Furthermore, while both certification schemes address several SDGs in general, these relate — in terms of biodiversity — mostly to SDG 15 on 'Life on land' that aims to 'ensure the conservation, restoration and sustainable use of terrestrial and inland freshwater ecosystems and their services, in particular forests, wetlands, mountains and drylands, in line with obligations under international agreements' and 'promote the implementation of sustainable management of all types of forests, halt deforestation, restore degraded forests and substantially increase afforestation and reforestation globally'.

6.2 General theory of change of SFM standards

The input of VSS-organisations consists of incentives to develop a market-based mechanism for ensuring sustainable production and consumption. Their output is the formalisation of these intentions into a set of principles, criteria and indicators laid out in standards containing the norm, certification and (third-party) verification procedures for checking compliance, market logos for labelling, and procedures to engage stakeholders in the control and review process. The outcomes of these norms, procedures and actions can be measured by the uptake of the standards and application of improved management practices by forest owners and timber producers. Indicators for outcome are for example the amount of certified wood and the total area of production forests that have been FSC or PEFC certified, after compliance to the standard has been verified. The impacts of effectively implementing the standards should result in positive biodiversity outcomes such as a reduction of forest degradation (avoiding biodiversity loss), and an increase in forest species (relative to more conventional wood production methods). The non-conversion criterion for plantations in FSC should result in avoided deforestation.

Principles and criteria for maintaining and enhancing forest biodiversity
The FSC and PEFC criteria can positively impact biodiversity conservation and even restoration in managed forests. The FSC standard (V5-2; FSC 2015) principle 6, for example, contains necessary actions for monitoring, maintaining and conserving, and even restoring environmental values, such as threatened species and habitats. Principle 9 mentions that a precautionary approach should be taken to identify and maintain parts of the forest management unit that are considered High Conservation Value Areas. Furthermore, forest plantations with usually low biodiversity values, can only be accepted for certification when they are not established on previously natural forest land (using a cut-off threshold year of 1995). PEFC adheres to similar principles and criteria, such as: Criterion 5.1.11 'Conversion of forests ……etc, including conversion of primary forests to forest plantations, shall not occur …'; Criterion 5.4.1 'Forest management planning shall aim to maintain, conserve and enhance biodiversity on ecosystem, species and genetic levels and, where

appropriate, diversity at landscape level'; 5.4.2 'Forest management planning, inventory and mapping of forest resources shall identify, protect and/or conserve ecologically important forest areas…'; and Criterion 5.4.13 'Standing and fallen dead wood, hollow trees, old groves and special rare tree species shall be left….' (see PEFC ST 1003:2010 for the complete text).

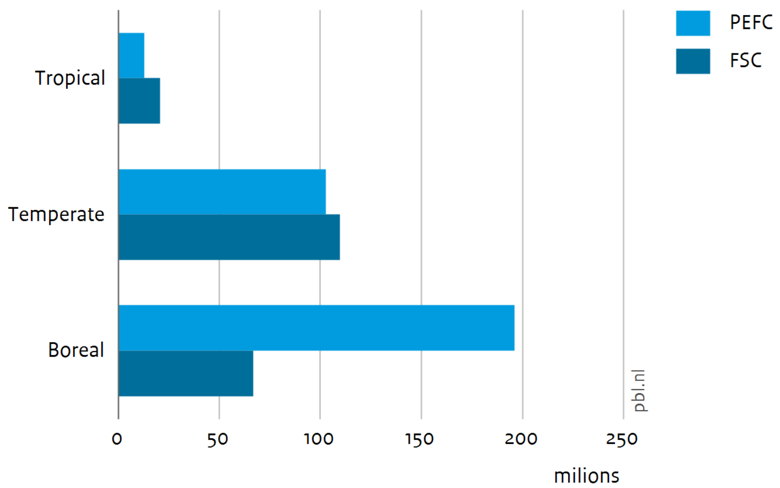
6.3 Assessing outcomes and impact in terms of the 3 assessment levels (3p)

6.3.1 Level 1: Global reach

Data from different sources and years had to be combined for this assessment. The data in this section was initially based on the State of Sustainability Initiatives 2014 and 2015 (Potts et al., 2014; Lernoud et al., 2015), and updated with more recent data sources on certified areas in different countries (obtained from FSC and PEFC websites, 10/09/2019). By 2019 FSC had 199 million hectares under certification and PEFC 311 million hectares, totalling 510 million hectares. Based on joint research, PEFC and FSC found that in mid 2018, over 86 million hectares (20.3%) were double certified[36]. So the actual global certified area corresponds to about 420 million ha, making up 10.6% of the world's forest area (based on World Bank 2016 data of 3.996 billion hectares total forest area[37]). Attributing the certified areas to climatic and forest types is needed for quantifying the certification impacts on biodiversity, as the effects of improved management practices differ between forest structures and forest management types.

Figure 6.1

Distribution of FSC and PEFC certified areas over climatic regions, 2019



Source: Data from FSC and PEFC websites 2019; processing PBL

[36] Based on an estimation by FSC and PEFC https://storage.googleapis.com/pefc-platform/pefc.org/media/2019-04/baecf2a2-144e-4c47-a24b-7a5df7ba2a24/bc3a2d7d-78d4-5863-b11c-d363da8ec380.pdf

[37] Extracted from https://data.worldbank.org/indicator/AG.LND.FRST.K2?view=chart

The areas covered by FSC and PEFC were aggregated to world regions and main climatic forest biomes. Certified forest areas were in 2019 mainly located in the boreal (52%) and temperate (42%) zones, while tropical regions only accounted for 6.6% of the total certified area. Forests under PEFC were predominantly found in boreal areas (63%: mainly Canada, Russia, and Northern Europe), whereas FSC is mostly found in temperate forests (56%: mainly Ukraine, United States, Central and Western Europe, and South America). Almost 11% of the FSC certified area is in tropical regions, while for PEFC certified areas this is only 6.5%.

The next step for the level 1 assessment is determining the management practices for forests in different climatic regions. Management in different forest types differ due to differences in forest structures (heterogeneity) and historical development of silvicultural practices. Selective logging for example, mostly takes place in natural mixed tropical forests with a heterogeneous population structure, while rotational felling is mostly practised in semi-natural and regrowth forests in the temperate and boreal zones with a more homogeneous structure. Artificial plantations are established all around the world. In a previous analysis it was determined that FSC is mainly implemented in natural and semi-natural/mixed forests, while plantations are much less present in the FSC coverage (van Oorschot et al., 2015). Data on PEFC certification per forest management type is not available, but since PEFC has a historical focus on conventional forestry in Western countries, it is assumed that most of PEFC certification takes place in semi-natural forests with rotational felling and natural regrowth, and in plantations where regrowth is done by planting and by using intensive management practices.

With the data on coverage of climatic regions, data on the distribution over different forest types and management systems (Arets et al., 2010), and a few assumptions[38], an estimate of areas under each type of forest management for the year 2019 can be given (see Table 6.1).

Table 6.1
Estimation of certified forest areas (million hectares) distributed over forest management types in 2019.

Certified area (million hectares)	Tropical	Temperate	Boreal	Total
Natural forest with selective logging	22	35	-	58
Semi-natural forest with rotation felling	-	88	219	307
Plantation forests	5.6	53	-	59
Total	28	177	219	423

Different sources of information for different years were combined to construct this distribution (regional management types from Arets et al., 2010; country level data obtained from FSC and PEFC websites; climatic distribution analysis by PBL)

6.3.2 Level 2: Change in management practices

To know the actual possible impacts of forest certification, we need to know the changes in management practices that take place under such certification schemes. These management practices were partly introduced above, but are further explained here.

[38] From the FSC and PEFC websites, data on concession size per individual country were downloaded, and countries were allocated to different climatic zones. In Arets et al 2010, the distribution of forest management types per climatic zone were assessed, and these percentages were applied to the 2019 FSC and PEFC area data. Data from different years were combined here, which leads to uncertainty.

Different mechanisms of change and corresponding impacts can be distinguished in forest certification. The type and intensity of management have a great influence on the biodiversity value of production forests (Alkemade et al., 2009; Burivalova et al., 2014). High biodiversity is found in protected and minimally used natural and secondary forests. Less high values, but still relatively good, can be found in multiple use, secondary or semi-natural forests (Putz et al., 2008).

Sustainable Forest Management practices as prescribed in certification standards can have impacts on biodiversity that include both on-site and off-site effects. The induced changes have different levels of magnitude, probability and certainty, and are described below.

1. Most natural and mixed forests in the tropics are exploited by selective logging. This means that specific individual trees of a commercially valuable species are logged. A characteristic feature is that wood production per hectare is low, and that the forest is degraded through careless logging practices and that damage is caused to the standing forest. In a case study on logging in Congo, a biomass damage to harvest rate of 1.75 to 3.0 was found (Brown et al., 2005). The damage rate usually becomes higher when the logging intensity increases (Picard et al., 2012).

By applying so-called Reduced Impact Logging (RIL) practices, several silvicultural techniques are practised to avoid collateral damage of logging and of removing wood from the forest. The RIL practices can benefit the regrowth capacity of the forest (recovery), the maintenance of healthy populations of commercial valuable tree species (sustainable forest exploitation), and reduce damage on commercially unimportant species (reduced degradation). Studies in Southeast Asia, Africa, and South and Central America have indeed documented that the negative impacts of selective logging on residual stands and soils can be substantially reduced through implementation of improved logging (Putz et al., 2008).

2. In **semi-natural secondary forests** that are widespread in the temperate regions (e.g. Europe, North America, China), the most common conventional production system is **rotation forestry**, where large areas of forest are cleared and regrowth is assisted by silvicultural management practices. This type of forestry is distinguished from artificial plantations by the used species (native species versus species exotic to the region) and the intensity of management (intense versus very intense). In a broad literature review, van Kuijk et al., (2009) found that, in spite of the large variety in responses between species, the different management practices in semi-natural temperate forests associated with certification appear to benefit biodiversity in managed forests. This holds for practices such as seed tree retention, establishing corridors connecting forest patches, maintaining riparian buffer zones, and set-aside practices (see under 4).

3. An essential criterion for certifying **forest plantations** is the requirement that a plantation may not be established by **converting natural forests**. When this happens, the original biodiversity will be lost, leading to forest degradation. It is this practice that can be avoided by adhering to the non-conversion criterion. Establishing plantations on degraded lands can then be a form of forest restoration. The on-site biodiversity effect of establishing a plantation will then depend on the former land use. We can now assume that when plantations are established on abandoned agricultural lands, MSA biodiversity values will rise, using the MSA values of different land-use types from Alkemade et al., (2009).

4. During the process of becoming certified, it is obliged to undertake **identification** and protection of High Conservation Value Areas (HCVA), and subsequently excluding these forest areas from

wood production (set-aside policy). In the definition of what is a valuable area, biodiversity is a prominent aspect next to other forest values like community needs, and providing locally important ecosystem services.

5. Another effect of forest plantations is the potential land sparing effect from **more efficient production**. Plantations have a high productivity, and this form of intensified wood production concentrates the production function on a smaller area, so less natural forests will have to be taken into production ('avoided loss'), or semi-natural forest under production management are allowed to recover to a more natural state ('regrowth' or 'restoration'). In a cost–benefit analysis it was shown that plantations in tropical South America can generate the same wood revenue as from natural forest exploitation, but on only 20% of the area (Arets and Veeneklaas, 2014). In Southeast Asia, plantations can produce the same revenue on about 40% of the area (conventional forestry practices are more intense in this region). However, there is no information available on land-use changes and potential sparing effects related to plantation establishment on a regional scale. So the potential effects of the sparing mechanism have a very low certainty; much depends on complementary forest governance for the forest that will no longer be needed. Therefore, this mechanism will not be estimated here in a quantitative way, as there are no clear indications of a direct certification effect.

6.3.3 Level 3: Biodiversity impacts of improved forest management practices

Now that the forest areas where different forest management practices in climatic regions are known (level 1, see Table 6.1), and the different management change mechanisms (due to forest certification criteria) are identified, we can estimate the potential impacts of certification on forest biodiversity. We estimate the differences in biodiversity between conventional practices and responsible practices as prescribed in standards for SFM. A first estimate for this positive certification impact can be made by using the Mean Species Abundance (MSA) biodiversity indicator. This is a common practice in long-term scenario comparisons with the IMAGE-GLOBIO model framework (e.g. PBL, 2010 and Kok et al., 2018).

To quantitatively assess the positive impact of certification, we took a two-step approach. First, the loss of biodiversity that would have happened under conventional forestry practices is calculated. The MSA indices for conventional forestry (Schipper et al., 2016) are multiplied with the areas for each forest management type (Table 6.1). This gives the impact weighted area sum, which is a useful basis for comparing different management options. After the loss under conventional management is quantified (Table 6.2), the positive effects of different management practices are estimated, thus partially mitigating conventional losses (Table 6.3).

Table 6.2
Theoretical loss of forest biodiversity in forest areas under certification, as it would be caused when these areas are managed by conventional logging practices

Areas of managed forests, and the remaining MSA under different management practices (cf Schipper)					
Forest Management	Residual MSA (index)	Certified area (10e6 ha)			**Totals**
		tropics	temperate	boreal	
Selective logging	70%	22	35	-	**58**
Rotation forestry	50%	-	88	219	**307**
Plantations	30%	5.6	53	-	**59**
Totals		**28**	**177**	**219**	**423**

Biodiversity loss in the areas that are certified, assuming conventional management					
Forest Management	MSA loss (index)	MSA loss x area (10e6 ha*MSA)			**Totals**
		tropics	temperate	boreal	
Selective logging	-30%	-6.7	-10.6	-	**-17**
Rotation forestry	-50%	-	-44.1	-109.3	**-153**
Plantations	-70%	-3.9	-37.1	-	**-41**
Totals		**-10.6**	**-91.8**	**-109.3**	**-212**

This loss is partly avoided and mitigated by applying SFM measures. The theoretical conventional loss serves as a benchmark to calculate the positive certification impact. Top half of the table is in million hectares, while the bottom part is the multiplication of areas times the MSA loss for the corresponding management practice. NOTE: MSA itself is an index on relative loss compared to the undisturbed reference, so formally dimensionless. However, we included the word MSA in the unit here to make clear these areas are quality weighted areas.

The estimate of impacts of conventional forestry in areas that are under certification (Table 6.2) serves as a benchmark for the assessment for the estimation of the added biodiversity by certifying forest areas (see for further explanation of MSA and application in scenarios Alkemade et al., 2006, and Schipper et al., 2016). The loss under conventional management would results in a total amount of 212 million ha*MSA in a forested area of 423 million ha. This means that on average about 50% of the original biodiversity is lost when conventional forest management is applied, and biodiversity friendly practices are not practised. Most of the loss can be seen in rotation forestry in the temperate and boreal regions, due to the large area where this management type is practised. Moving on from this step we can now include the positive effects of the different certification mechanisms.

1. Selective logging in Natural and recovered mixed forests
In Schipper et al. (2016), the residual biodiversity in lightly or selectively used forests is found to be 0.7 MSA. This means that about 70% of the species populations that can be found in unaltered natural forests are still present, and that 30% has been lost. There are hardly any experimental studies available that compare conventionally logged forests with certified selectively logged forests. But there is a lot of research available on specific silvicultural techniques that are prescribed in certification standards (van Kuijk et al., 2009). For the sake of simplicity, SFM in selective logged tropical forests is equated here by implementing RIL and other supporting techniques (Peña-Claros et al., 2008).

In a meta-analysis based on >100 publications (Putz et al., 2012), several positive effects were found when mitigating collateral damage and practising more sustainable silvicultural treatments, like: sustained timber yields after the first harvest cycle, although at a somewhat lower level; retaining three quarters of stored carbon in once-logged forests; and keeping 85% to 100% of mammal, bird, invertebrate and plant species after logging.

Further, a literature review by (Arets and Veeneklaas, 2014) contains several estimates on the mitigating effects on damage by applying RIL techniques. In South American tropical forests, logging damage to forest biomass is on average 32% for Conventional Selective Logging (CSL), and only 17% for RIL. In South East Asia, where higher logging intensities are common, this is on average 54% for CSL and 28% for RIL. Although there is discussion about the effectiveness of RIL techniques, we conclude that there is enough evidence available for attributing positive on-site biodiversity effects to RIL. Based on the review, and assuming that biomass loss in mixed forest is a proxy variable for biodiversity, a modest positive effect of 15 percentage points (assuming a range between plus 10 to plus 20 percentage points) higher biodiversity is attributed to RIL compared to conventional selective logging, roughly halving the biodiversity loss caused by selective logging.

2. Improved rotation forestry in semi-natural secondary temperate forests

In Schipper et al. (2016), the residual biodiversity in these intensively used forests is found to be 50% MSA. Under level 2 it has been discussed that there are several positive effects noted from improved silvicultural techniques for rotation forestry (van Kuijk et al., 2009). Further, Karmann and Smith (2009) also mention positive improvements in temperate forests as a result of FSC certification. These general positive impacts were confirmed in interviews with scientific experts, forest certifiers and forest managers (Zagt et al., 2010), and in the review by Di Girolami (2019, 2023). However, no quantifications could be derived from these qualitative literature reviews, so rough assumptions are needed for a first-order estimate. The evidence base provided in van Kuijk (2009) is clear enough to award the certified practices with a relatively modest positive potential biodiversity effect. As a rough approximation, we assume a positive effect of 15 percentage points (uncertain), reducing the impact of conventional rotation forestry by about a third.

3. Non-conversion criterion for plantations

There is no information available on the type and frequency of land-use changes related to plantation establishment and certification, apart from local examples (Reynolds et al., 2011). Therefore the positive effects of the non-conversion criterion is also a potential effect. Different types of land-use changes may take place. A change from intensively used cropland (10% residual MSA) to a forest plantation (30% residual MSA) would result in an absolute biodiversity increase of 20 percentage points. As a very rough estimate we assume here that half of the certified plantations are established on abandoned agricultural lands. And the other half is assumed to be established on abandoned rangelands for which the average MSA value (over plants and animals see Schipper et al., 2016) is about equal to that of plantations, so no net impact will appear here. This assumption leads to an average increase of plus 10 percentage points (uncertain).

4. Set-aside practices for forest areas identified as having a HCV status

From a limited number of case studies on forest plantations (NGPP and WWF 2009), set-aside percentages have been derived, containing the amount of area that is put under conservation management in a forest concession area. So by setting aside, part of the forest concession area is excluded from exploitation. The cases showed that between 5% and 40% (uncertainty range) of the total area of a forest management unit or concession falls under the HCV definition (High Conservation Value). This wide range is caused by differences in forest structure and history, local conservation priorities, and different high-value definitions. This ‘avoided loss’ effect is in the end quite modest, as it only applies to a limited part of the certified and managed forest area. To give an estimation of this effect, we assume that otherwise lost biodiversity by establishing plantations is not taking place in the area that is set-aside. No information was available on set-aside practices in other forest management types.

5. Sparing effect of putting plantations in place
The last effect of sparing forests from exploitation or even avoiding deforestation cannot be calculated here, as it is not a direct result of certifying forest concessions. It has to be captured by additional policies on regional land-use planning and governmental conservation policies. An indication of this effect can be obtained by performing global long-term land-use scenarios (see Kok et al., 2022).

6. Combined effects of different improvements in forest management
When we apply these estimated positive impacts of practices promoted under certification to the estimated areas of each management type in **Table 6.1**, the total potential biodiversity effect of forest certification can be calculated by multiplying the estimated positive MSA effects with the respective areas. In this way, we can compare the positive biodiversity effects of certified sustainable forest management with the hypothetical biodiversity loss under conventional forestry (**Table 6.2**). A full uncertainty analysis is at this point unfeasible, as there is no information on probability distributions of each change mechanism.

In Table 6.3 the combined positive effects of different practices in SFM are presented and taken together. The different effects of on-site improvement in rotation forestry delivers the largest positive effects (measure 2), due to the extensive areas where rotation forestry is practised. For this effect, the certainty is low, as the actual improvements are not known. The quantified effects of RIL are the most certain, but show up as a low contribution due to the small amount of certified area that we currently see in the tropics. Additional effects like establishing plantations on former abandoned agricultural lands, and setting aside areas of special conservation value deliver a modest positive effect (measures 3 and 4). Not surprisingly, most of the total calculated effects are seen in the temperate and boreal zones, and much less in the tropics.

The combined effect of improved forestry management (lower half of Table; measures 1 to 4) is 64 million ha·MSA. So implementing measures for improved forest management is potentially able to avoid about a third of the loss from conventional logging (64/212 = 30%).

Table 6.3
Estimated positive mitigating effects of management practices that are promoted under certified forest management

Areas for which specific improved management practices applies						
Forest management types			Area (10e6 ha)			
			tropics	temperate	boreal	
Selective logged forests			22	35	-	
Temperate and boreal forests			-	88	219	
Plantations			5.6	53	-	
HCVA area in plantations			1.3	11.9	-	

Positive impacts of SFM practices in different forest Management types						
Potential certification impacts	Positive impacts		Impact x Area (10e6 ha*MSA)			
	min	max	tropics	temperate	boreal	Totals
1. RIL application	10%	20%	3.4	5.3	-	8.7
2. Improved rotation forestry	10%	20%	-	13.2	32.8	46
3. Plantation establishment	0%	20%	0.6	5.3	-	6
4. HCV set-aside in plantations	5%	40%	0.9	8.3	-	9
5. Avoided deforestation (sparing)			not quantified as it is indirect			-
Totals			4.8	32.2	32.8	64
			45%	35%	30%	30%

*The total positive effect of several measures is calculated as 64 million ha*MSA, which means that the loss under conventional practices of 212 million ha*MSA is reduced by almost 30%.*

6.3.4 Level 4: Implementation effectiveness

By performing a Systematic Literature Review (SLR) Di Girolami (2019) assessed the environmental impact of forest certifications around the world, both FSC and PEFC. Her review assessed 883 possible titles as a starting point, but only included 29 true impact studies, which cover 49 assessments on flora, fauna, and ecosystem services, representing about 13 million ha — or about 3% — of certified forests around the world. A wide range of impacts were found in this body of literature. Indicators for environmental impact in these studies relate to floristic composition, tree species diversity, wildlife species, biomass, dead wood, forest disturbance, carbon stock, forest areas for conservation, and reduced deforestation among others. For biodiversity specifically, one study (Kalonga et al., 2016) looked at the two FSC certified community forests, and compared them to open access forests and state forests in Tanzania. It showed that the richness, diversity and density of adult tree species in the FSC certified forestry communities was significantly higher than in the other areas.

Now 32 of these 49 environmental impact assessments report positive results, 15 show no impact, while 2 report a negative impact. These figures imply an 'implementation effectiveness' of about 65%, which seems quite a good performance for forest certification. However, one must be aware of a 'positive bias' in such scholarly literature. The positive bias was tested for in a second step where only assessments from this SLR with the highest quality and rigour scores based on quasi-experimental design, are taken into account (Di Girolami et al., 2023). At this point the success rate drops to 50%. Consequently, we conclude from this SLR that about half of environmental impact assessments on forest certification report at least some positive results. However, one should be careful in generalising these findings beyond the rather low number of studies covered by this SLR. In addition, the degree of positive environmental impact could not be derived from this SLR.

6.4 Discussion and conclusion

Potential positive effect of forest certification

This calculation exercise using different estimations on the positive effects of measures that are prescribed in certification systems shows that a substantial amount of the loss due to conventional forestry can be avoided in certified forests. The loss in the total area under certification (423 million hectares) would amount to 212 million ha*MSA (212/423=-50% loss) when this area would be managed conventionally. By applying SFM, a positive impact of +64 million ha*MSA was estimated. This means that the loss that would have occurred of -50% is now brought back to only -35% ([212-64]/423=-35%). This avoided loss of 15 percentage points gives certified forests a 30% relatively better biodiversity status compared to conventional logging (64/212=+30%).

Uptake of certification in tropical regions is limited

The findings for level 1 raise questions about the promises of certification, and objectives being achieved. Only 7% of certification is taking place in countries and regions containing tropical forests that have initially raised concerns about deforestation and forest biodiversity loss in Africa, South America and Asia (van Oorschot, 2015). Despite most deforestation and illegal logging having occurred in tropical forests over the last three decades, only 3 countries in the top 15 of certified countries contain tropical forests (Australia, Brazil, and Malaysia). This shows that forest certification is predominantly concentrated in Northern developed economies and largely not happening in the countries that see more drastic declines in forest area.

Forest protection or Forest exploitation — Use it or lose it?

Avoiding deforestation is for a large part a matter of value creation. There is discussion whether it is best to keep forests untouched (with high biodiversity values, but with limited economic existence value), or whether it is better to allow sustainable exploitation and keeping the forest in a good condition. A comparison between a protected forest and FSC production forests in Mexico showed that generating income from sustainable logging is a good alternative to protected forests, for which financial sources were difficult to find (Hughell and Butterfield 2008). Deforestation and forest fires were much more frequent in the protected forest, where informal use and illegal practices frequently occurred. The study did not address the presence and protection of specific species, and the effects of sustainable forest use on benefits for the local population. This example shows that sustainable forest management is a way to preserve the forest and protect it from degradation and conversion. A supporting finding for this strategy is that sustainably managed forests contain a significant amount of forest biodiversity, estimated at 85% by Putz et al. (2008). Next to creating economic values, there is also the argument of societal values. The economic value of agricultural practices is much higher than of forestry, which is in favour of conversion. But an important characteristic of sustainable forestry is that it not only gives economic value to forests, but also maintains several societal benefits (van Oorschot et al., 2015). Certification can therefore also be regarded as a market mechanism that provides incentives to conserve societal good standing, by preventing forests from being converted to agriculture. An important conclusion is that sustainably managed forests have an important conservation function, and that this strategy of sustainable forest use is complementary to strict protection of forests. Putz et al. (2008) therefore conclude that 'selectively logged forests retain substantial amounts of biodiversity, carbon, and timber stocks', and therefore, this 'middle way' between deforestation and absolute forest protection deserves more attention and acknowledgement from researchers, conservation organisations, and policymakers.

The certification paradox

There is a paradox in forest certification (Potts et al., 2014), as much is expected with regard to improving management and making an impact on both forest biodiversity and also working conditions and income for forest smallholders and workers. This promise on making a change is not always seen in practice, as certification is known to focus on those managed forests where management is already close to the required criteria. As demand for certified wood rises, it is expected that the certification process will gradually shift towards practices where more change is needed. However, this depends on the availability of additional funds, to cover the higher costs to make management in line with requirements. This paradox is the reason we call the estimated positive effects on biodiversity 'potential impacts', that do not necessarily materialise in practice. The estimated positive effects should therefore be seen as an optimistic representation of certification effects. A possibility to overcome this barrier for creating impact is to raise the bar for all producers, so as to help producers perform at a distance from the requirements (PWC and IDH, 2012). The existence of national forest governance is seen as an important aspect of the enabling conditions for successful certification (Cashore et al., 2003). Special EU supporting programs in the form of bilateral agreements are now set up to help tropical countries to improve their forest laws and forest governance, and to build capacity to enforce them. This will help to create a level playing field for all producers on legal aspects, and not just for the ones seeking easy certification. These Voluntary Partnership Agreements (VPAs) are implemented for countries that are relevant for EU imports, amongst others Cameroon, Central African Republic, Ghana, Indonesia, Liberia, and the Republic of the Congo.

Monitoring need

In the above discussed and quantified effects, comparisons are used between conventionally managed and sustainably managed forests. But this gives a somewhat hypothetical and optimistic idea of forest change brought about by the certification process. To obtain a reliable image of actual certification effects, data on pre-certification forest biodiversity status and on effects of implemented management improvements is needed. However, there is to our knowledge no publicly accessible monitoring system that tracks and reports the measured effects at this moment. The absence of sector wide representative performance and impact monitoring makes it at present impossible to produce a reliable estimate of the share of certified areas where improved forest management has induced positive biodiversity effects.

7 Integrated Landscape Management

Indigenous people and local communities have been managing landscapes for thousands of years. Such socio-ecological production landscapes, mosaics of habitats and land uses, have been shaped over the years by human-nature interactions. They have proven to be, when managed well, sustainable and able to maintain biodiversity while providing ecosystem services for human wellbeing, and can often be considered cultural heritage (Bhakta et al., 2016; Gu and Subramanian, 2012). Over the last centuries many approaches towards practices, and versions of, land management have emerged and existed. These span various land uses, in a range of ecological conditions, and with different types of decision-making and institutional arrangements. These approaches can be brought together under the umbrella term Integrated Landscape Management (ILM). Integrated Landscape Management is described as the management of a landscape that involves collaboration among multiple stakeholders, and has the purpose of achieving sustainable landscapes (Meijer et al., 2018a). It reconciles economic, social and environmental concerns within a holistic framework and potentially offers an important opportunity for both the provision of ecosystem services and human development (Whitbread-Abrutat, 2012).

In the last few decades there has been increased international attention for landscape restoration, and a landscape is a suitable scale for management as well as for combining multiple global goals through restoration. This momentum is heavily driven by non-state actor initiatives such as: the Landscapes for People, Food and Nature Initiative (LPFN), the Global Partnership on Forest and Landscape Restoration (GPFLR), the Global Restoration Initiative (consisting of WRI, IUCN and other partners) (Arts et al., 2017), and the research by for instance the Satoyama Initiative on Socio-ecological production landscapes (SEPLs). Activities on the ground range from many small scale, bottom up initiatives such as those in the Sahel, to large scale, top down government driven restoration programs. The Landscapes for People Food and Nature (LPFN) initiative notes that more than 80 communities of practice have been documented managing landscapes to achieve positive outcomes across multiple sectors, including participatory watershed management, pastoral community conservancies, ecosystem management, forest landscape restoration, climate smart territorial development, indigenous landscape management, agricultural green growth, climate-smart landscapes, food–energy water nexus systems, and city-region food systems (LPFN, 2015).

The sustainable use of landscapes through the appropriate management and restoration of landscapes is highly relevant to biodiversity governance, as agriculture, habitat conversion and deforestation are the largest drivers for biodiversity loss. Restoration activities often restore or rehabilitate lands, and at the same time might halt and/or prevent further degradation and conserve the landscape as it is. The added value of ILM for biodiversity is the integrated approach towards preserving and restoring ecosystems and its functions, in which conservation and ecological restoration efforts are in balance with other land uses, including sustainable agriculture and agroforestry to support economic development (Arts et al., 2017). In recent years, sustainable land management has been included in ambitious cross-sectoral national targets and gained a more prominent position on the international agenda. Examples are the Bonn Challenge, and the New York declaration of Forests (Arts et al., 2017).

As integrated landscape management can take many forms, we focus our case (following the work by N. Estrada-Carmona et al., 2014; Milder et al., 2014) on integrated landscape projects, programs, platforms, initiatives, or sets of activities that: (1) seek to conserve biodiversity or ecosystem services; (2) work at a landscape scale; (3) involve inter-sectoral, multi stakeholder coordination or alignment of activities, policies, or investments on various scales; and (4) are highly participatory.

Integrated Landscape Management can be positioned as a hybrid among ICIBs as government authorities are often part of these initiatives. Multi-stakeholder, including national government, and sometimes international, partnerships set up these integrated landscape management projects, but many bottom-up local initiatives exist as well, which can also include local initiatives supported by international NGOs.

7.1 Motives, goals, targets

Motives: Large-scale forest clearing and the expansion and intensification of agriculture are major causes for biodiversity loss and ecosystem degradation. The restoration and sustainable use and management of landscapes can play a significant role in conserving and restoring multiple ecosystem functions in parallel with preventing further loss. It reconciles economic, social, and environmental concerns within a holistic framework and potentially offers an important opportunity for both the provision of ecosystem services and human development (Whitbread-Abrutat, 2012).

ILM has the **goal** to conserve and restore ecological integrity as well as improve human well-being through multifunctional landscapes. In practice, it follows this overall goal, but can also have different specific goals, varying according to which natural elements, uses, and stakeholders a landscape has.

As such ILM can also have different objectives depending on the area, but in general can add to the realisation of existing multiple nationally and internationally agreed **targets**, such as the Kunming-Montreal targets of the CBD, particularly Target 3 which 'aims to ensure and enable that by 2030 at least 30% of especially areas of particular importance for biodiversity and ecosystem functions and services, are effectively conserved and managed, and integrated into wider landscapes, seascapes and the ocean'. And also Sustainable Development Goal 15 'Life above land' — to 'protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss'.

7.2 Theory of change: from input to impact

As introduced above, integrated landscape initiatives address the need to contribute to international sustainability agendas, with integrated approaches to several forms of landscape management to provide benefits to both people and nature. Integrated landscape initiatives benefit from having a helicopter view of the environmental and socio-economical context of an area, and therefore balance environmental targets with other sustainable development goals (i.e. not sharing or sparing, but where applicable, both), which reflects the **input**. Multi-stakeholder landscape scale initiatives have been around for a long time in the form of landscape architecture, spatial planning, watershed management and similar planning and management approaches.

Where governance of land has been poor, bottom-up landscape management has been initiated. As such, integrated landscape management is a collection of efforts, with a variety of goals for different local stakeholders. Integrated landscape initiatives often combine several land uses' within a mosaic landscape, a mixed landscape, or include for instance ecological corridors in production lands (pastures, agriculture, plantations). From these approaches, or **outputs**, multi-stakeholder landscape scale projects, or programs arose that combine existing efforts related to e.g. forestry, agriculture, nature conservation and sustainable livelihoods, under one landscape umbrella in a more integrated manner. In practice, this included, with and by a variety of stakeholders, new local regulations, change of management practices, spatial planning and behavioural changes. As an outcome, once owned or co-managed by oneself, local stakeholders in general feel more responsible for the resource and have a stake not to lose the benefits they reap from those landscapes, leading to more sustainable landscape management practices, conserving natural elements of the landscape and substantially reducing (or halting) degradation. The added value for biodiversity is the consideration for indirect effects of activities within a landscape on others, the reduction of pressures on its natural elements or areas (incl. biodiversity) and about increased conservation and connectivity. Impacts therefore consist of improved livelihoods and ecosystem condition and hence also positive biodiversity outcomes.

7.3 Past performance: assessing outcome and impact

For assessing the outcome and impact of integrated landscape initiatives, we use data from a collaborative group of researchers who have together published a White Paper and related reports based on a survey and stocktaking of Integrated Landscape Initiatives (ILIs) across the world (Estrada-Carmona et al., 2014; García-Martín et al., 2016; Milder et al., 2014; Zanzanaini et al., 2017). Results include mostly qualitative data about integrated landscape initiatives (ILIs) in Latin America and the Caribbean (Estrada-Carmona et al., 2014), in Europe (García-Martín et al., 2016), in South and Southeast Asia (Zanzanaini et al., 2017), and in Africa (Milder et al., 2014). Outcomes of these studies mention biodiversity conservation as one of the impacts of the efforts carried out by these initiatives.

We have used the **African data set** (Milder et al., 2014) to test our methodology on ILM and to give insight on the on-the-ground impact of these ILIs on biodiversity. Milder et al. (2014) used online keyword searches and interviews to identify and screen ILIs throughout Sub-Saharan Africa. They included ILIs initiated by various actors and donors, as well as those with traditional, locally developed landscape management systems addressing challenges such as land degradation, climate change, population growth, or major external investments. Milder et al. (2014) collected basic information on each longlisted initiative to assess its suitability for final selection, according to ILI criteria. One representative of each selected initiative was then invited to complete a survey, consisting of checkbox, multiple choice, and open-ended questions. The survey focused on characterising each ILI and its context: (1) landscape location, size, population, and land cover; (2) basic information about the initiative; (3) activities and investments within the initiative, in agriculture, conservation, human livelihoods, and institutional planning and coordination; (4) participation of stakeholder groups and sectors; (5) (self-reported) outcomes of the initiative; (6) most and least successful aspects, and (7) basic information about the respondents and their organisation. They analysed the results (excluding missing values or incomplete responses) to generate descriptive statistics (see Milder et al., 2014 for more details).

There were two reasons to look into the African data set only: 1) due to time constraints, we decided to focus on a smaller data subset, and could not dive into possible differences between the regional studies, which were not all conducted by the same researchers or at the same period of time; and 2) Milder et al. (2014) already found that in Africa, overall biodiversity protection was among the most reported impacts, and was mentioned in more than half of the initiatives from their study.

7.3.1 Level 1: Global reach

The first step of analysis is to take stock of the total number of integrated landscape initiatives across the globe, and the size covered on the ground by their projects. We found that, according to the series of studies on ILIs, about 1500 ILIs were identified (of which they analysed 423 ILIs sub-selection based on whether survey responses were complete and matched their ILI criteria):

- 382 ILIs throughout Latin America (including Mexico, Central America, and South America) as well as the major Spanish-speaking Caribbean jurisdictions of Puerto Rico, Cuba, and the Dominican Republic (N. Estrada-Carmona et al., 2014).
    - 104 ILIs (out of 173 responses) were analysed.
- 338 ILIs throughout Europe (33 countries; incl. United Kingdom, Belgium, Germany, Italy, Spain, the Netherlands, France, Sweden, Romania, Austria, and Estonia) (García-Martín et al., 2016).
    - 71 ILIs (out of 86 responses) were analysed.
- 649 ILIs throughout South and Southeast Asia in Bangladesh, Bhutan, Cambodia, China, (Southwest China and Tibet), Laos, India, Indonesia, Malaysia, Myanmar, Nepal, Pakistan, the Philippines, Sri Lanka, Thailand, and Vietnam were invited for the survey (Zanzanaini et al., 2017).
    - 161 ILIs (out of 227 responses) were analysed.
- 105 ILIs throughout Sub-Saharan Africa (survey responses only) (Milder et al., 2014)
    - 87 ILIs were analysed (in 33 countries in Sub-Saharan Africa; mostly in Kenya, Ethiopia, South Africa, Democratic Republic of Congo, and Uganda).

As mentioned before, we will zoom into the African data set as published in Milder et al. (2014), for our further analysis. For level 1, determining the total amount of hectares of ILI with a positive impact on biodiversity, a sub-selection of the data set by Milder et al. (2014) was made in several steps, to make sure to only include ILIs from his database that (also) focus on biodiversity (since these ILIs do not necessarily aim to impact biodiversity, we will select only those who do have motive and/or investments to do so), based on the survey questions and answers:

Table 7.1
Sub selection process of integrated landscape initiatives with a positive biodiversity impact in Africa.

87 Integrated Landscape Initiatives in Africa		
Additional selection based on survey responses	Included	Excluded
Response ticked from list: 'Conserve biodiversity' to the question 'What were the main motivations for the initiative? In other words, what problems or challenges was the initiative trying to solve, or what opportunities was it trying to realise? (Check all that apply)'	68	19
Response ticked from list: To the question: 'Please tell us about the major activities, investments, or other changes that were included as part of the initiative.' Investment in forestry, conservation, and natural resource management: New protected areas established; New management plans for existing protected areas; Other new reserves or community based conservation areas (including areas that allow sustainable harvest and use of natural resources); Other community based natural resource management activities; Improved forestry management; Watershed management program or activities (e.g. restoration of riparian areas) From agricultural investments: Adoption or expansion of agroforestry; Implementation of laws or incentives to reduce the environmental impacts of agriculture; Promotion of native food species and agrobiodiversity	65	3 (these did indicate such activities being done simultaneous to the ILI, but not by the ILI itself)
ILIs with participation by government are included, but ILIs that ticked no non-state actors (e.g. NGOs, Women's groups, farmer associations etc.) to the question: 'Which of the following types of groups have participated in designing or implementing the initiative?' nor mentioned such stakeholders to the question 'Which organisations lead the initiative?', and thus only national or international governments, have been excluded.	65	0
TOTAL level 1 ILIs	65 ILIs	22 excluded

Based on the Milder et al. (2014) database.

Respondents were asked how large the landscape area of the ILI was. **This amounted to around 360 million ha for the 87 ILIs from the survey, of which 330 million ha for the 65 ILI landscapes in which biodiversity conserving or restoring activities by these initiatives were implemented (expected biodiversity increase)**. More detailed numbers cannot be given due to some ambiguity in the responses. Important to keep in mind is that these areas can be quite large, and positive biodiversity outcomes do not necessarily take place in the whole landscape.

7.3.2 Level 2: Change in management practices

Our second level analysis is to provide data on the coverage of areas positively affecting biodiversity in the areas of the landscape projects mentioned in the survey, as well as the kind of activities undertaken. Preferably, we would need the specific or mosaic of activities implemented and their corresponding hectares for the ILIs, but this was unfortunately not reported in the survey. However, the respondents of the survey did give their perception of the resulting impact of these landscape initiatives. Note that survey respondents were invited based on their familiarity with each initiative and the landscape in which it is embedded, and responses are thus self-reported. For more on this, please see our 'discussion' sub-section.

Table 7.2
Indication of how many initiatives had intended and which had indirect impacts on biodiversity

65 Level 1 ILIs (expected biodiversity outcome)	Included	Excluded
Responses ticking 'Overall biodiversity of the region was better protected' or 'Rare, threatened, or endangered species were better protected', to the question: 'Effects on conservation and ecosystem services'.	47	18
Which indicates 47 out of 65 initiatives (72%) have had an intended positive impact on biodiversity.		

22 Excluded level 1 ILIs (no expected biodiversity outcome)	Positive Outcome
Excluded Level 1 ILIs who did report on having an positive outcome on biodiversity (indirect effect) (survey responses ticking: 'Overall biodiversity of the region was better protected')	5
Which indicates 5 out of 22 non-biodiversity ILIs (23%) had an indirect positive impact on biodiversity.	

Based on the Milder et al. (2014) database

Respondents were asked how large an area has been directly affected by the initiative's activities, programs or policies (remember that the total project area, which we see as 'areas with expected biodiversity impacted', of the 65 ILIs which implemented biodiversity conserving or restoring activities, from level 1 was around 330 million ha). This amounts to 200 million ha affected area (actual biodiversity increase) for the 47 ILI landscapes (in which 240 million ha was expected). In addition to these 47: an area of about 320,000 ha from the re-included 5 of 22 excluded ILIs (representing an area of which reported indirect effects on biodiversity. **Thus, a positive impact on biodiversity has been reported for 72% ILIs (47 out of 65) on 61% (200 out of 330 million ha) of their area**. In addition, 23% (5 out of 22) of the other non-biodiversity ILIs reported a positive impact on biodiversity. More detailed numbers cannot be given due to some ambiguity in the responses.

7.3.3 Level 3: Biodiversity impacts under ILM

Level 3 analyses would provide quantification of actual biodiversity impact for the activities identified in level 2, expressed in Mean Species Abundance (MSA, indication of the naturalness of the landscape), which we would determine as relative MSA increase, instead of relating it to a pristine ecosystem baseline. Without the specifics of what was done in the project mentioned in the survey, and the amount of hectares, the total MSA impact could not be quantified.
Here, we would like to point to a case study in Tanzania by Meijer et al. (2018b) who calculated MSA values for an integrated landscape scenario made by stakeholders. They found that when looking at the whole landscape the current MSA value was 63% (thus 37% of the original biodiversity, in MSA, has been lost compared to an undisturbed pristine situation). In 2030, none of the scenarios is able to completely halt the loss of biodiversity, however where under a Business as Usual (BaU) scenario the MSA strongly declines to 38%, in the Integrated Landscape scenario MSA is maintained at 58%. Taking into account the projected increase in population, much biodiversity loss is caused by the conversion of natural to agricultural areas, hunting, and climate change. Nevertheless, in the Integrated Landscape scenario, 80% of the loss in biodiversity can be prevented compared to the BAU scenario (Meijer et al., 2018b).

7.3.4 Level 4: Implementation effectiveness

Based on these results, we are able to deduce that most integrated landscape management initiatives did experience improvements in biodiversity protection and conservation. The impacts for such initiatives were positive, and supported a more integrated approach of these initiatives.

Critically, it was interesting to witness that there were several other initiatives that without having biodiversity as main focus, were able to positively contribute towards outcomes for biodiversity conservation.

Despite these results, the implementation effectiveness of ILM with regards to the case study, cannot be declared as indisputably conclusive. There are some challenging limitations that precludes this study from doing so which are discussed in the 'Discussion and Conclusion' section. Summarily, there are a few barriers to overcome in order to understand the implementation effectiveness of ILM. Firstly, as ILM is a very large umbrella for several initiatives of varying nature, there is a lack of descriptive detail of activities on the ground. Secondly, activities under the ILM umbrella are highly variational in scope, scale, and duration with projects ranging across multiple geographies and timelines. This makes a direct one-on-one comparison of cases impossible to execute. We discuss these limitations further in the next sub-section.

7.4 Discussion and conclusion

7.4.1 Methods and data

First of all, the most important limitation of this data set for the purpose of this case study is that there are no detailed descriptions of what kind of activities were actually implemented on the ground, nor is there a subdivision of the impacted area of the ILIs related to such specific activities. However, to our knowledge also no other databases are available for this, so this was the best data for us to work with.

Level 3 analysis would have been partly possible if we would have had at least land use data. The complete MSA indicator is based on various modules determining impacts from land use, fragmentation, infrastructure, nitrogen deposition, hunting and climate change. To fully assess the impacts from landscape scale ILI interventions (e.g. promoting mixed or mosaic landscapes, improved landscape planning and management, connectivity etc.) more spatially explicit information would be needed. Given that land use change is a major cause of biodiversity loss, just having insights on these MSA impacts would already have provided a limited assessment on the effects of ILIs. Besides, for integrated landscape management it would do better justice to these initiatives to also look at indicators of ecosystem services and agrobiodiversity. Furthermore, please note that some projects that were part of the survey were really short in duration (2 to 4 years) whereas others are planned to continue until 2050, and it is thus unclear how much monitoring of the beneficial effects of these activities has (already or since) been done.

Also, the survey responses are self-reported by people that have been involved in these initiatives, as respondents were invited based on their familiarity with each initiative and the landscape in which it is embedded, indicated by their residency or long-term work in the landscape. They were also selected on having a broad understanding of activities and issues spanning multiple scales and sectors in the landscape. Of course, intentional or unintentional biases in self-reported responses on ILI outcomes are a limitation of using this methodology. This will certainly have had an influence on the estimated benefits of the programme, though it is hard to say how much (see Milder et al. (2014), for more details on how have attempted to mitigate these limitations).

7.4.2 Further research

For future research on non-state initiatives on Integrated Landscape Management (ILM), it would be recommended to do a comparable case study analysis instead. ILM cases do not fall under one institutional umbrella and are, by definition, highly diverse in terms of activities implemented. Perhaps a comparable case study analysis in combination with empirical field data would offer more insight into possible biodiversity impacts (as well as other ecosystem services) on the ground following landscape level activities being implemented. Furthermore, we left out addressing agricultural or crop biodiversity from the scope of this study, although this was part of the survey responses. It would be an interesting aspect of biodiversity for a follow up study, and links with for instance the non-state Slow Food movement.

8 ICLEI

Globally, urban populations have grown rapidly since 1950, having increased from 751 million to 4.2 billion in 2018, representing about 55% of the world's population (UN-WUP, 2018; UNDESA, 2018). It is expected that by 2030, urban areas are projected to house 60% of the global population, and about 70% in 2050 (UNDESA, 2018; CBD, 2012). As cities adapt to accommodate an increasing population, urban expansion is inevitable. According to the Cities and Biodiversity Outlook of 2012 over half of the projected urban area in 2050 was yet to be built. This urban expansion, or sprawl, puts increasing stress on natural resources, land and ecosystem services (CBD, 2012). As for impacts to biodiversity specifically, many cities are situated in biodiversity rich areas such as floodplains. As rapid urban expansion occurs, the subsequent habitat loss has resulted in severe consequences, especially in cities bordering biodiversity hotspots (CBD 2012; the Nature Conservancy, 2018).

In addition to increased urban populations, climate change will add pressure on cities as well. Effects of climate change such as sea level rise and changes in temperature and rainfall will have negative impacts both on biodiversity as well as the services provided by ecosystems that are essential for human survival. Local governments of cities vulnerable to these effects need to address these in both integrated climate change mitigation and adaptation strategies, as well as biodiversity strategies and action plans (UN-Habitat, 2012). Understanding urbanisation trends in the face of ongoing climate change is crucial to the implementation of the 2030 Agenda for Sustainable Development, especially Sustainable Development Goal 11: 'to make cities and human settlements inclusive, safe, resilient, and sustainable' (UNDESA, 2018).

The 'Global 200 ecoregions' (conservation priorities due to having high species richness or endemism, or being under high degrees of threat), 'Biodiversity Hotspots' (>1,500 endemic vascular plants that have lost more than 30% of their original natural habitat), 'Alliance for Zero Extinction' (AZE; where the only known population of a particular species exists) and IUCN's Key Biodiversity Areas (KBA meet >1 of 11 criteria, e.g. threatened or geographically restricted biodiversity, irreplaceability) identify roughly the same biodiversity hotspots forecasted to have significant urban growth: central Mexico, the southern coast of Brazil, and southern China. AZE's are also disproportionately found on islands, such as in Madagascar, Indonesia and Papua New Guinea. Other concentrations of urban-impacted KBAs are in Europe, the Caribbean, the western coasts of South America, Taiwan, Korea, Japan and in Africa — most commonly along coastal regions such as the Mediterranean, the Gulf of Guinea, East Africa and the coast of South Africa (the Nature Conservancy, 2018).

The relation between cities and biodiversity is not a negative one only. Cities provide a variety of opportunities for biodiversity and ecosystems. Especially because they are often located in biodiversity hotspots, they can, if managed well, support biodiversity by acting as refuges for species whose habitats have been destroyed; acting as socio-ecological systems where new habitats and species communities can develop; and by providing local ecosystem services such as noise reduction, absorption of air and water pollutants. In some areas, urbanisation even reduces pressures on land, allows considerable regrowth and increases biodiversity. Cities can be very rich in biodiversity and remarkable amounts of native species diversity can be found in and around for instance Singapore, Rio de Janeiro, Chicago, Berlin and Stockholm (presentation Thomas Elmqvist Urban Nature Forum, 2012).

It is estimated that about 5.2 million ha of KBAs could be lost to urban growth by 2030. Protecting these could be a significant first step toward mitigating the impact of urban growth on biodiversity (The Nature Conservancy, 2018).

At international and European level specifically, we find a range of targets and policies on nature (e.g. Sustainable Development Goals; EC, 2013), but non-state level action could play an important role as well, especially when it comes to urban biodiversity conservation. As more cities become aware of the environmental and sustainability challenges they face, a number of opportunities that urban nature can offer to deal with the broader challenges are also increasingly palpable. This has stimulated a rise in the number of cities and wider urban municipalities adopting sustainability policies for Nature-Based Solutions (NBS) that might (directly or indirectly) benefit urban biodiversity. There are several international non-state organisations (e.g. ICLEI), which act as facilitators and catalysers in the development and implementation of urban biodiversity policies.

A comprehensive global database of all cities that have biodiversity policies currently doesn't exist. ICLEI — Local Governments for Sustainability — is a leading non-profit global network of cities and towns, committed to building a sustainable future, and their member list thus gives an overview of part of the cities active with biodiversity policies. To aid cities in incorporating nature into their plans, ICLEI has created 'Cities With Nature' (with TNC and IUCN). This is a global platform for cities and other sub-national governments that recognises and enhances the value of nature in and around cities. It builds on earlier ICLEI initiatives such as the Local Action of Biodiversity (LAB) initiative, the Communication, Education, and Public Awareness (CEPA) program; the LAB Wetlands program; the Integrated Action for Biodiversity Project (INTERACT-Bio), and the Urban Natural Assets (UNA) program. It draws also from lessons learned under the Cities' Biodiversity Index (or Singapore Index) (the Nature Conservancy, 2018).

This case study aims to provide estimates of the number of ICLEI cities with urban biodiversity policies and strategies, the urban green infrastructure that is implemented under such policies; and finally, the estimated actual impact on biodiversity that urban green infrastructure has on the ground. Urban Green Infrastructure (UGI) refers to a strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services (incl. biodiversity). In urban areas, UGI might consist of parks, gardens, green walls, green roofs and much more. These are part of an interconnected network and are delivering multiple ecosystem services (EC, 2013). This combination of quantitative and qualitative information on the impact of urban strategies on biodiversity would then add to our wider analysis of how non-state initiatives benefit biodiversity.

8.1 Motives, goals, targets

The **motives** for ICLEI members' urban biodiversity policies vary, but the main driver is often similar: expected benefit from the improvement of urban green infrastructure on biodiversity and ecosystem services to deal with urban challenges. The city can be seen as a socio-ecological system with the four categories of ecosystem services being present: (I) Provisioning services include food and fresh water supply; (II) regulating services include air quality regulation, pollination and carbon sequestration; (III) cultural services include recreation, stress release and education; and (IV) habitat services (including biodiversity[39]) (Connop et al., 2016; TEEB, 2011; MA, 2005). Urban biodiversity policies generally cover a variety of these services, and often support the socio-ecological system as a whole.

The theoretical **goal** of ICLEI members' biodiversity related efforts would be to conserve or increase biodiversity, while maximising ecosystem services provided for human wellbeing (e.g. climate change mitigation), through, among other things, Urban Green (and blue) Infrastructure (UGI).

In practice, **goals** for re-naturing cities depend on the specific initiative. Relevant to this study are those goals, which directly relate to biodiversity are of interest, such as, outlined in for example Local Biodiversity Strategy and Action Plans (LBSAP) or similar documents. Examples are, 'Preserving and enhancing the natural heritage of the city and preventing species and habitats from disappearing' (Barcelona City Council, 2013), and 'Reservation, conservation, recovery and protection of biodiversity — To increase the biodiverse vegetation coverage area in the city' (Sao Paulo City Hall, 2011).

Of the plans identified and reviewed in this study, very few set out detailed quantitative targets. However, several global goals are relevant for urban biodiversity plans, in particular the Kunming-Montreal **targets** of the CBD, particularly Target 12 which aims to increase the area and quality and connectivity of, access to, and benefit from green and blue spaces in urban and densely populated areas sustainably.

8.2 Theory of change: from input to impact

Input is the intent of a city or other local government to produce and commit to a strategic plan, which aims to maximise biodiversity and ecosystem services, delivered by urban green infrastructure. This intent to produce a strategic plan can be encouraged, and facilitated by international non-state partnerships. Output, in this case, would take the form of a strategic plan, a city community platform or other local policy directed at renaturing cities and urban biodiversity, which is open to the public. At the very minimum, strategic plans should contain goals, but ideally also include targets and a timeframe for achievement. Outcome entails behavioural change such as the implementation of the strategic plan. This might include creation, protection and restoration of urban green spaces, management of invasive species, but also citizen's management of private and public greens. The resulting impact is the protected or enhanced biodiversity and other ecosystem services in support of biodiversity within cities or urban spaces.

8.3 Past performance: assessing outcome and impact

As said, the first level of analysis assesses the overall number of ICLEI cities that have policies on urban biodiversity, and how many ICLEI cities are involved with international non-state partnerships or initiatives. Unlike the other case studies, level 1 here does not refer to hectares of

[39] Conceptually tricky: *Biodiversity could be seen as the underlying factor to providing services OR as one of the several services. More on this under 'discussion'.*

green space, as no comprehensive database on that is available. As far as comparability of this case study to the others in this report, the total amount of hectares is of relatively low importance when comparing urban initiatives to state-led initiatives. Within cities, the percentage coverage of areas of urban green infrastructure would be relevant, so that is what this case study aims to do under level 2.

8.3.1 Level 1: Global ICLEI area

The first step was to take stock of the total amount of ICLEI membership across the globe. From the ICLEI website, the exact number of active cities involved in ICLEI seems to be layered. We found that some cities that are involved in ICLEI programmes (such as the ICLEI LAB programme) are not listed as members. ICLEI itself states: 'ICLEI Members are at the heart of the network. They steer the direction of our work, shape our strategy and play a central role in our governance structure. The wider ICLEI network of more than 1,500 engages directly in any one or more of our projects, partnerships and initiatives and actively uses our various platforms and tools. Our growing network reaches: 1750+ cities, towns and regions.' As we will focus on the activities of a collaborative, we will consider all cities that are core ICLEI members and listed in their membership database. This list currently has **868 ICLEI members** (as of 21 August 2019, ICLEI website), which includes a range of urban area types such as cities, towns, municipalities, villages, metropolitan areas, counties, states, provinces and associations of local authorities.

For the purpose of our level 2 to 4 analyses, having a comparable list of urban areas is needed to quantify the impact of this collaboration. To filter out in some way the cities from the ICLEI membership base, which also contains for instance counties, provinces and villages, we chose to cross reference ICLEI membership with a list of cities in the world as explained hereafter. What defines a city and its size is not universal, and we used the World Urbanization Prospects (UN WUP) of the United Nations Population Division which lists urban agglomerations with a population of >300,000 (UN, 2014 and 2018), as was the most comprehensive list available to us. In 2018 specifically, the UN WUP data contained 1900 cities and urban agglomerations of over 300,000 inhabitants (WUR, 2017; UN WUP, 2018).

Of the 868 ICLEI members, 292 (33.6%) are cities >300,000 inhabitants (UN WUP, 2018). The other two thirds consist of urban area types such as smaller cities, towns, municipalities, villages, metropolitan areas, counties, states, provinces and associations of local authorities. Interesting to note is that **217 (25%) out of the 868 ICLEI members are located in the United States. Moreover, 292 of the ICLEI members represent 15% of the 1900 cities >300,000 inhabitants** (as of August 2019, according to members listed on ICLEI website).

Although with the UN WUP data this study will focus on cities with populations >300,000 inhabitants, it is good to note that Fuller and Gaston (2009; in a study on relationships between urban green space coverage, city area and population size across 386 European cities) found that green space provision within a city is primarily related to city area rather than the number of inhabitants. Compact cities (small size and high density) show very little green space allocation per capita. As cities grow, landscape quality outside formal urban green infrastructure (e.g. street plantings size, composition and management of backyards and gardens) will increasingly determine interactions between people and nature (Fuller and Gaston, 2009).

8.3.2 Level 2: Change in management practices

In our second level analysis, the idea would be to provide data on coverage of areas of urban green infrastructure in ICLEI member cities, and a description of the kind of activities undertaken, reported as implemented or planned in urban biodiversity policies. Moreover, under the second level analysis we will also be looking at changes in management practices in cities to explore the biodiversity impacts it could potentially have.

Due to the lack of benchmark data, providing an estimation of coverage area of activities proved infeasible within the scope of this case study:

- Although biodiversity is a key point of ICLEI's agenda, it addresses sustainability in a broader sense. There is currently no database of specific biodiversity inclusive strategies of ICLEI's members. Therefore, for this report we looked into specific ICLEI biodiversity programs: LAB and INTERACT-BIO. ICLEI started LAB, a biodiversity project with 21 pilot cities, to kick-start LBSAP (Local Biodiversity Strategy and Action Plan) pledges for the CBD. However, information was not consistently recorded across cities, and the on-the-ground data on what was implemented remained scarce with final reporting on LAB outcomes unclear. The more recent INTERACT-Bio initiative by ICLEI focuses on fast-growing city regions in Asia, Africa, and South America and supports coordinating efforts to mainstream biodiversity and ecosystem services into core subnational government functions. This programme started in 2018 and reporting on what was achieved was not complete when this case study was conducted. Also, not all cities involved in the ICLEI initiatives were listed as ICLEI members. This further complicates building a consistent data set.
- We used Google, Google scholar and literature recommended through expert consultation and snowballing from other articles to find more on ICLEI members' city biodiversity websites, programmes and case studies mentioned. Some individual documents were useful, but together did not form a consistent, comparable data set.

Based on the available data it was therefore not possible to define a comprehensive quantitative biodiversity impact for our level 2 analysis. More on this lack of data, in section 8.4.1.

We can illustrate what kind of efforts could be part of cities' biodiversity strategies, as well as an example case study to show how to possibly quantify these activities on biodiversity impact in levels 3 and 4. As our methodology has a strong focus on area based data (hectares), we will mention only area based activities here. In practice, these fall under Urban Green Infrastructure that address biodiversity.

<u>Possible strategies for area based biodiversity restoration and conservation in urban areas:</u>
There is some debate on which strategies in urban planning have the most impact on conserving biodiversity in cities and natural resources under pressure of urbanisation (Botzat et al., 2016). As already mentioned under level 1, some argue that more compact cities could slow down the processes of urbanisation and biodiversity loss outside cities (Botzat et al., 2016). Others however found that dense cities could result in smaller, more segregated green spaces that might hamper urban people's access to ecosystem services and could have limited contribution to biodiversity conservation (Botzat et al., 2016).

1. *Urban Green infrastructure — area based measures for biodiversity*

The implementation of re-naturing or re-greening city policies will influence, preserve or expand the Urban Green Infrastructure (UGI). The implementation of UGI has been promoted as a strategy to improve human health and wellbeing and adaptation to climate change, especially in Europe. However, the contribution of UGI to urban biodiversity conservation is largely unrecognised. Butt et al. (2018) found that although most city climate change adaptation plans are committed to green infrastructure goals, only few contain specific intentions to enhance biodiversity. This limited acknowledgement within climate adaptation plans might point to underestimation of the contribution that green infrastructure could provide for achieving both outcomes, if explicitly planned in an integrated manner (Butt et al., 2018). Providing more green space to urban citizens is one suggested policy strategy to address both environmental and socio-economic challenges. For example, the United Nations' Sustainable Development Goals (UN, 2015), particularly goal 11 'sustainable cities and communities' calls for more green spaces in cities. Another important policy that stimulates UGI to incorporate biodiversity goals is the EU policy: Green Infrastructure strategy (EC, 2013).

2. *Urban Green Infrastructure (UGI) to connect cities to surrounding landscapes (reconnection)*

Both species richness and composition change along the urban gradient. Lowest species richness found in the city centre. Thus, to preserve biodiversity in expanding cities, urban planning needs to focus on preserving remnant natural habitat (as most current land developments remove natural vegetation during construction). For species composition, most notable is that non-native species become more common toward the urban core, whereas native species that avoid urban areas such as large predators and forest-interior (especially insectivorous) birds that disappear quickly in the initial stages of suburban encroachment. The latter can only be avoided by retaining large areas of native habitat and reducing human persecution of these species. On the other hand, certain mammals and birds that are mainly adapted to forest edges and open areas, flourish in suburban habitats, especially older sections where ecological succession has advanced and produced extensive re-vegetation (McKinney et al., 2002).

3. *Conservation of green elements in expanding cities (avoided loss)*

Since 1950, urbanisation and respective growth of existing urban areas has been increasing (see Table 8.1). According to an analysis of 202 European cities (Kabisch and Haase, 2013), residential urban areas seem to increase regardless of population growth or decline, resulting from increases in households and the number of smaller households demanding larger living spaces. To preserve biodiversity, natural vegetation in central urban areas should be protected, while ensuring spatial distribution that enables the connectivity between them. For instance, so-called inner urban forests are given a new role, and their connection to surrounding suburban forests represents an important link between urban areas and other forests on a broader landscape level (Hladnik and Pirnat, 2011; Pirnat and Hladnik, 2016).

Table 8.1
World, number of agglomerations 1950–2020 (UN, 2018)

Size class of urban settlement	**No. of Agglomerations**						
	1950	1975	2000	2005	2010	2015	2020
10 million or more	2	4	16	20	25	29	34
5 to 10 million	5	14	30	36	39	45	51
1 to 5 million	69	145	325	343	380	439	494
500 000 to 1 million	101	223	396	456	510	554	626
300 000 to 500 000	129	258	524	591	645	707	729
TOTAL	306	644	1,291	1,446	1,599	1,774	1,934

This increasing sprawl on the margins of cities in the past few decades has driven policies to increase urban density in many developed nations. While this slows land conversions, compaction has a significant impact on urban green space provision. Maintaining green space quantity and quality in the face of increasing urbanisation is therefore a pressing global challenge (Fuller and Gaston, 2009). For instance, for bird biodiversity, contrary to what we mention under point 2, Callaghan et al. (2018) study on 112 green spaces from 51 cities across eight countries found that distance from the city centre and distance from the coast were not significantly related. Rather, the size of a green space was the most important predictor of bird biodiversity, and therefore also the most important factor for increasing bird biodiversity and for mitigating loss from urbanisation.

4. *Creating Urban Green Infrastructure (UGI) as Nature Based Solutions*

Non-biodiversity specific sustainable urban development programmes can have positive effects on urban biodiversity as well. For instance, the carbon® Climate Registry (cCR) is one of the most widely used reporting platforms in the world for cities, towns and regions. It provides self-reported data on local and regional climate action. About 15% of these, (half of which are from the Global South) have reported to have either 'completed' or are 'in progress' of completing their Climate Adaptation Plan. In these, 'preserving/improving ecosystems and biodiversity'' is among the most reported co-benefits from their adaptation actions (ICLEI, C40, 2018). Vice versa, a broad range of urban green spaces could support not only biodiversity but broader living conditions by increasing a range of ecosystem services, benefitting physical and mental wellbeing and social cohesion (Botzat et al., 2016).

Example Cape Town level 1 & 2

The City of Cape Town (CCT) in South Africa signed the Durban Commitment in 2008, and was a member of the ICLEI Local Action for Biodiversity (LAB) programme. As part of LAB, they selected five 'biodiversity implementation projects' to address key biodiversity challenges in the city, and produced several policy documents such as a Biodiversity Report, a Local Biodiversity Strategy and Action Plan (LBSAP), a Biodiversity and Climate Change Assessment Report, and a Biodiversity and Communication, Education, and Public Awareness (CEPA) Report (Elmqvist et al., 2013). There are three spheres of government involved with biodiversity within the City of Cape Town (Rossouw et al., 2013): 1) South African National Parks (SANParks), responsible for the management of the Table Mountain National Park, 2) CapeNature, the provincial conservation authority, responsible for the implementation of biodiversity conservation in the Western Cape Province and the management of provincial nature reserves of which there are two within CCT, and 3) the City of Cape Town (CCT) itself, responsible for the management of 31 City owned conservation areas.

The City of Cape Town adopted a city policy, the Cape Town Bioregional Plan, in July 2015. This bioregion encompasses the City of Cape Town metropolitan area of 246,000 ha (26% urban development, 35% agriculture, 39% natural remnants cover). It indicates Critical Biodiversity Areas (CBAs) and Critical Ecological Support Areas (CESAs) (CCT, 2015; Holmes et al., 2012). Under this policy, the BioNet is implemented, which includes all priority natural and semi-natural wetlands and rivers. The BioNet remains the core of the CCT's strategic biodiversity conservation. In 2009, the CCT protected areas only accounted for 2,294 ha in and overall (including other protected areas in Cape Town) only 34.1% (29,003 ha) of the BioNet was conserved. In 2018 this has grown to 64.8% (54,745 ha), moving close to their 2022 target of 65.5% (CCT, 2018a; CCT, 2018b).

Besides the protection, management and acquisition of conservation areas, other biodiversity projects by the City of Cape Town include:

1. The Invasive Alien Species Coordination Programme: Invasive alien species (IAS) are the second biggest threat to the biodiversity of the City of Cape Town. Accordingly, an IAS strategy has been

developed to address the current challenges and to provide the framework for improved IAS management (CCT, 2009).
2. The Core Flora Sites form an important component of the BioNet lowlands. In 1999, 7703 ha of Core Flora Sites were non-conserved, 1694 ha conserved. In 2018, only 3278 ha remained non-conserved, with conserved sites doubled to 3467 ha and an additional 2013 ha in progress (CCT, 2018a).
3. Other conservation activities included: the Ecological Restoration Prioritisation Project (restoration of priority areas of the remaining habitat in the city), re-vegetation of wetlands under the Peninsula Rehabilitation project; specific species (flora and fauna) protection projects; and several programmes on fire (wildfires, ecological burn, brush pile burning) (CCT, 2018a).
As the initial and post-implementation status of these areas is difficult to determine, defining MSA values related to the activities done here are not obvious. Also, no MSA values are available yet on invasive species. Furthermore, it is unclear to what extent these numbers overlap (e.g. how much overlap there is between invasive species management in river catchments and the protected area sites).

8.3.3 Level 3: Biodiversity impacts under ICLEI practices

Level 3 analysis would provide quantifications of actual biodiversity impact for the activities identified in level 2, expressed in MSA (which we would determine as relative MSA increase, instead of relating it to a pristine ecosystem baseline). Taking note of the fact that urban biodiversity policies often aim for supporting ecosystem services or even to play a hotspot or corridor function for biodiversity in the wider landscape, their policies usually do not regard it as feasible or an objective to ‘bring back’ and restore biodiversity to the extent possible. This makes using MSA as an indicator tricky for this case study (which serves as a proxy for ‘naturalness’ and refers back to the pristine ecosystem in comparison to current conditions, Alkemade et al., 2009). If biodiversity in cities increases as result from certain measures, this biodiversity is not necessarily reflective of the original ecosystem, but very much depends on the activity that was implemented: conserving and restoring areas in the city with natural vegetation or ‘re-greening’ cities with natural vegetation would increase ‘naturalness’, but cities are full of exotic biodiversity as well (e.g. tropical garden plants, invasive wild species, botanical gardens etc.). For these native natural areas, MSA could still be used looking at the relative change of MSA, for example from 0 or 0.1 for severely urbanised areas to an MSA of 0.5 for a park under restoration with native species. At the time of writing, however, MSA values for UGIs had not been developed, yet.

8.3.4 Level 4: Implementation effectiveness

While the momentum of ‘green’ and ‘smart’ cities seems to be growing, it is good to note that in our search for coherent reporting on both plans and implementation, we found that a lot of programmes on green/renaturing cities have been launched over the last decades but have never been properly reported on, or officially closed. Hence, it was not possible to give an estimate here in how many of the cases the initiatives were actually successful in increasing biodiversity.

What we can say in more general terms about implementation success is that, for the correction factor of area based biodiversity/nature conservation in cities, it is important to take into account ‘connectivity’ and spatial planning: a mosaic landscape with large patches and some finer-grained areas within it would provide both, as a combination of patches and corridors are key for increasing connectivity and preventing fragmentation (Forman in UN Habitat, 2012). Another factor to include would be restoration quality related to management and species composition. Pesola et al. (2017) found a negative relationship between biomass and biodiversity. Above-ground biomass and biodiversity relationships in urban forests seemed dynamic across different stages of development.

In summary, this chapter on ICLEI aims to contribute towards the issue of biodiversity conservation by expanding the scope on how biodiversity may be perceived. Moreover, this chapter also proposes the inclusion of additional measures for biodiversity other than just MSA.

In the context of the implementation effectiveness, this chapter highlights three reasons why it could not be conducted holistically. These are discussed in detail in the next section. Firstly, the scope of this chapter was limited by language restriction within literature review which was limited to English thereby missing out on several literature on the subject. This was further compounded by the lack of distinct demarcation between city and general urban areas which made sampling for this chapter challenging. Secondly, there doesn't seem to exist a database of ICLEI activities that could be collated, analysed, and compared for biodiversity measures. This also made data availability a problem. Thirdly, in order to understand biodiversity impacts that ICLEI initiatives might possess, it is critical to understand and include the definition of biodiversity from an 'urban areas' perspective wherein, the biodiversity and ecosystem services are measured using different measurable variables.

8.4 Discussion and conclusions

8.4.1 Method and data

An aspect that will have affected our results is that our literature searches (see level 2) were done in English. It is very likely that there is relevant academic literature as well as LBSAPs and similar documents which did not appear in this assessment due to search term bias and language issues. For example, there are indications that LBSAPs exist for several Chinese cities among those in the 4+ million inhabitants category. These might very well have been missed in our English search. Future analyses on cities' LBSAPs for instance, should consider including more languages.

The UN World Urbanization Prospects defines cities by urban agglomeration. A downside to this is that it combines a number of urban areas and considers it as one. The Greater Tokyo Area, for example, also includes cities such as Yokohama (3+ million) and Kawasaki (1+ million). Using a city as a unit could possibly be more relevant when looking at urban green infrastructure and related policies, as this would delineate cities at the governing authority level, which is likely also where the jurisdiction of parties with strategic plans end. However, data for city proper was not available from a reliable source in a single format.

We found that currently a coherent overview of existing urban activities under ICLEI, cities' LBSAPs, City Biodiversity Index etc. does not exist and creating such a database will be complex. There are several initiatives under way to create such overviews, such as UB Hub (https://www.ubhub.org/) and the Urban Nature Atlas created under the Naturvation project (https://naturvation.eu/atlas and Almassy et al., 2018). Having these databases is crucial to do research on the extent of what cities, municipalities, and wider peri-urban regions are doing and can do for biodiversity, but also for a wider set of ecosystem services. Such databases and platforms are also important to showcase all that is already being done by local governments on reaching national, regional and international biodiversity targets, such as the Kunming-Montreal targets of the CBD. Making this visible, would not only give limelight to the sub-national efforts that are ongoing, but might also serve as a stimulant to act and opportunity for knowledge exchange for other local governments.

What do we mean by biodiversity? This is an especially sensitive discussion in urban areas, where each square metre is under pressure of delivering services to the urban inhabitants. When it comes to biodiversity, this raises the discussion: is biodiversity one of the habitat services, under the umbrella of ecosystem services? Or is biodiversity the enabling environment for these ecosystem services to manifest? In their systematic quantitative literature review of 200 studies on perception and valuation of biodiversity in cities, Botzat et al. (2016) also found that there is are large knowledge gaps in what we know of biodiversity in cities, and found four biases in current literature: 1) strong focus on urban forests and parks while important informal green spaces are largely neglected; 2) biodiversity is mostly addressed at the ecosystem scale, while biodiversity at community or gene levels are underrepresented, although important for conservation; 3) almost no studies on the cultural diversity of urban residents, which carries the risk of policies neglecting that varying demands of cultural groups on the design, management or biodiversity of green spaces; and 4) most literature covers temperate zones, while urban growth hotspots are underrepresented. The last is especially worrying, as Butt et al. (2018) found that areas where urban population growth is expected to expand the most (Africa, Asia) were less likely to have climate adaptation plans in place, while biodiversity conservation could potentially have the greatest impact in these areas, through such climate adaptation plans. (Butt et al., 2018).

There is a lack of proper instruments to deal with biodiversity at the city level, and when they do exist, there is little coordination to put them into practice. Proper biodiversity indicators for cities are key but the results will take time to affect policy, and changing the planning process to incorporate biodiversity issues is still a major challenge (UNU-IAS, 2010).

8.4.2 Further research on urban biodiversity in the future

In addition to ICLEI members, other cities address biodiversity in their urban plans as well. Examples are Local Biodiversity Strategy and Action Plans (LBSAP) or documents of similar scope. LBSAPs can be considered the local variant of National Biodiversity Strategy and Action Plans as produced under the Convention of Biodiversity (CBD). LBSAPs are either separate biodiversity strategies or, to integrate urban biodiversity and ecosystem services into local governance, LBSAP elements are incorporated into overarching urban plans which can trickle down to guide sector-specific plans integrating biodiversity (CBD, 2012). Of the 1900 WUP database cities, we could find 77 cities that have produced and published an LBSAP or similar document (or are in draft stage of writing one), 6 additional cities signed the Durban Commitment, indicating the ambition to write an LBSAP.

It was beyond the scope of this study, but it would be interesting to also consider what can be said about urban biodiversity in the coming decades:

- The biggest urban growth is expected to be in small and medium-sized cities, not in megacities (CBD, 2012), so it might be interesting to do another study looking specifically into those smaller cities that are expected to grow much bigger in the coming decades.
- Assessing what ratio of cities with an urban biodiversity policy will have a positive effect on biodiversity requires establishing a benchmark and follow-up assessments. Many of urban biodiversity policies are relatively new, which suggests it may take some time before sufficient data is available. Moreover, a global assessment would require uniform data. The LBSAPs — or

cities from the Subnational Biodiversity Strategies and Action Plans (SBSAPs) list — (CBD website, accessed 27th Jan 2019) are a starting data set to indicate city biodiversity policies.

- The City Biodiversity Index (CBI), also known as the Singapore Index, is a self-assessment tool that was developed for national governments and local authorities to assist them in benchmarking biodiversity conservation efforts in an urban context and encourages cities to monitor and evaluate biodiversity conservation and enhancement more specifically (Kohsaka et al., 2013). The CBI could provide the benchmark needed to determine the proportion of urban biodiversity policies with a biodiversity impact in level 2 analyses. Currently, about 50 cities are in various stages of providing data for this index. However, very little public information is available, let alone a database, on the CBI of the cities that committed to reporting one (CBD, 2012).
- For future research, it would be interesting to compare cities' LBSAPs and urban biodiversity policies to the National Biodiversity Strategy and Action Plans (NBSAP) as published under the CBD. This could result from cities creating policies that impact biodiversity beyond national policy. To this end it would be interesting for further analysis to compare several LBSAPs with the relevant NSBAPs and determine whether local plans do indeed exceed national plans.
- Reporting on outcomes of projects would also be interesting for any patterns in those monitoring and evaluation reports: which kind of programmes did make it, what does the reality of local governance have to do with this (in which mandates, budgets, elections are different from national and global level governance?) etc.

9 Rewilding Europe

Humans have been shaping and altering landscapes for millennia, often, especially in the past decades, with detrimental consequences for biodiversity. At the same time, ongoing land abandonment (mostly in Europe) has made the restoration of degraded landscapes an increasingly important topic (Kuemmerle et al., 2008; Navarro and Pereira, 2015; Qeuiroz et al., 2014). Balancing these two issues, rewilding has become a rapidly emerging concept in restoration conservation.

Rewilding practices have been present for a long time. Examples of early 'rewilding' practices are for instance the reintroduction of wolves in Yellowstone National Park in 1995 (Fritts et al., 1997; Dobson, 2014), the 'Room for the river' plan in the Netherlands in 1996 through which floodplains were re-established and human activity was brought to a minimum (Drenthen, 2009), and the reintroduction of large grazers in the Oostvaardersplassen in the Netherlands in 1983 (Vera, 2009). The general goal behind rewilding practices is to restore natural processes with minimal human intervention (Navarro and Pereira, 2015). Furthermore, it is not only an important strategy to restore nature, but also lives up to meeting the demand of an urbanised society to experience wild and rough nature (Corlett, 2016). The rewilding movement is increasingly gaining attention and support, with some labelling it the emergence of a new environmental narrative coined 'Recoverable Earth' (Jepson, 2018).

Rewilding activities are often linked to the concept of landscape restoration (Drenthen, 2018). Whereas many other conservation efforts concern some kind of active management such as afforestation or agricultural intensification (Navarro and Pereira, 2015), rewilding allows for short-term interventions, such as the introduction of keystone species, but is then followed by passive management. Much debate still exists around both the term and the practices of rewilding, such as differences between the European and North American interpretation of rewilding (Oliveira-Santos and Fernandez, 2010; Martin, 2005; Hall, 2014; Corlett, 2016). The underlying goal of rewilding, in the European sense, is generally not to restore a previous landscape state, as is often the case in the North American interpretation, but to restore ecological processes and thereby return ecosystems to an autonomous state — a state in which they can function on their own. Jørgensen (2015) provides an overview of the interpretations of rewilding throughout scientific literature. She outlines six different uses; (1) cores, corridors and carnivores; (2) Pleistocene megafauna replacement; (3) island taxon replacement; (4) landscape through species introduction; (5) productive land abandonment; and (6) releasing captive-bred animals into the wild. In a response to Jørgensen's study, Prior and Ward, (2016) define rewilding as follows, thereby encompassing all these different views into one definition: *'a process of (re)introducing or restoring wild organisms and/or ecological processes to ecosystems where such organisms and processes are either missing or are 'dysfunctional''.*

Corlett (2016) further explains her view on the differences between rewilding, restoration and conservation translocation in which restoration is described as including 'restoring species composition, structure and function to an approximation of a historical reference system, as well as the less ambitious targets of reforestation, revegetation, rehabilitation and reclamation and the more human-focused approach of ecological engineering', conservation translocation as 'the movement and release of organisms for conservation reasons, including reintroduction and reinforcement, where the organisms are released within their indigenous range, as well as conservation introductions outside this range, to avoid extinction (assisted colonisation) or to restore ecological function' and rewilding as a combination of two different approaches; 'trophic rewilding, where the aim is to restore ecosystem functions by restoring top-down trophic interactions, and passive rewilding, where human interference is minimised from the start'.

Over the last decades, a growing number of initiatives have focused on restoring natural processes through minimal interventions or passive restoration, such as Rewilding Europe[40], Rewilding Great Britain[41], Dam Removal Europe[42], but also smaller initiatives focusing on single species or areas (e.g. zu Ermgassen et al., 2018; Sandom et al., 2014; Valdés-Correcher et al., 2018). However, overall global cooperative initiatives for rewilding do not exist. For this case study we therefore take Rewilding Europe as an example for the rewilding movement.

Rewilding Europe (RE) is one of the main International Collaborative Initiatives on Biodiversity (ICIBs) which has put rewilding practices on the international agenda. As one of the only institutionalised ICIBs on active rewilding interventions, Rewilding Europe acts not only through their own efforts, but also through their European Rewilding Network, connecting local activities to an international cooperative network. The idea behind Rewilding Europe is to combine the restoration of natural processes with broader collaboration with rural and urban communities. In this way, self-functioning ecosystems can be created along with public support for wild(er) landscapes, economic benefits from nature through, for example, tourism and various ecosystem services beneficial to both local communities and nature itself (Cerqueira et al., 2015). Since Rewilding Europe is currently the biggest organisation with the main goal of rewilding, and data on their practices has been made available, we will assess their impacts as an example of an ICIB in the context of rewilding.

9.1 Motives, goals and targets of Rewilding Europe

The **motive** for Rewilding Europe seems to come from the large-scale agriculture and human activity and its drastic effects on European landscapes. For Rewilding Europe, current abandonment of European agricultural land provides an opportunity to increase biodiversity, since these landscapes might otherwise lose their biodiversity benefits. Through rewilding these landscapes, natural processes can be restored and further degradation of ecosystems can be prevented. Next to this, Rewilding Europe aims to combine their rewilding efforts with 'reconnecting the wild(er) nature with the modern economy' and 'responding to and shaping cosmopolitan perceptions of nature conservation among European society' (Jepson and Schepers, 2016).

Rewilding Europe has the **goal** of creating autonomous landscapes, while at the same time improving the connection between modern society and nature. The goal is not to recreate the past, but to take inspiration from it. With that and with the inclusion of people, natural processes can be restored and wilder, passively managed, European landscapes can be recreated (Jepson and Schepers, 2016).

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[40] www.rewildingeurope.com

[41] www.rewildingbritain.org.uk

[42] www.damremoval.eu

Rewilding Europe works with 5- and 10-year objectives **(targets)**: (1) ‘creating enabling conditions and kick-starting the more natural functioning of nature across Europe’, (2) ‘ensuring the continued comeback of wildlife, including large herbivores, large carnivores and scavenger across Europe to service both nature as well as people’, (3) ‘demonstrating that rewilding generates new business opportunities, jobs and income for society, thereby creating an alternative and competitive form of land (and sea) use for local people, landowners and communities’, (4) ‘generating pride, public support, new partnerships and a more positive attitude amongst stakeholders for a Europe with much more wild nature, wildlife and wilderness’ and (5) ‘inspiring the scaling up and replication of the rewilding approach across Europe’ (Rewilding Europe, 2018a). Target (1) and (2) are of interest to this study due to their connection to a possible increase in biodiversity.

9.2 Theory of change? From input to impact

The **input** for Rewilding Europe was the establishment of the organisation in 2011 by four co-founders from WWF Netherlands, Wild Wonders of Europe, Conservation Capital and ARK Nature with the aim to ‘make Europe a wilder place, with more space for wild nature, wildlife and natural processes’ and at the same to ‘explore new ways for people to enjoy and earn a fair living from the wild’ (Rewilding Europe, 2018b). The **output** was the creation of nature restoration, or rewilding strategies. The idea behind Rewilding Europe’s practices is that natural succession often leads to continuous scrublands, followed by forest. Rewilding, however, restores natural processes that were eliminated by humans, such as the grazing of large grazers, and in that way creates an autonomous ecosystem with higher biodiversity (Rewilding Europe, 2018b). Furthermore, rewilding also provides ecosystem services such as the regulation of water, recreation services and the protection of soil and nutrients. The **outcomes** of Rewilding Europe’s practices are agreements with landowners. Rewilding Europe does not buy land to protect it, but makes agreements with landowners to protect their lands and carry out rewilding practices. Through working together with these landowners, Rewilding Europe is able to extend their practices without the need to buy land off other people, and to manage large areas of land in close collaboration with the stakeholders. Through working with the local people, Rewilding Europe further aims to facilitate nature in a modern society through letting people create businesses, jobs and employment from it. The **impacts** of Rewilding Europe’s practices are aimed at higher biodiversity values, while at the same time creating benefits for people.

9.3 Assessing the outcome and impact

Rewilding Europe is a relatively young organisation. The topic of rewilding is also a concept that only recently has been gaining more momentum. It therefore provides for an interesting initiative to analyse by the different levels in the assessment framework. The assessment provided interesting results for level 1, 2, 3, and 4.

9.3.1 Level 1: Global reach

The assessment framework resulted in several challenges for the case of Rewilding Europe, which could not (yet) be overcome. For level 1, we were able to obtain data from Rewilding Europe. This data still provided some insecurity. The current total targeted area of Rewilding Europe encompasses 4.97 million hectares; this is not all under Rewilding Europe’s management. However, some of this area is directly influenced by concrete actions on the ground such as the protection of landscapes through agreements with land owners, the introduction of large grazers, the reduction of poaching and the construction of wildlife corridors, activities that provide the possibility to be assessed using MSA values. While the majority of the 4.97 million hectares are suggested to be under influence of the wider impacts of the interventions, will be under influence of future interventions and some parts are impacted through other interventions (Wouter Helmer, pers. comm.). This difference consequently also provides a challenge in quantifying the impact of Rewilding Europe's practices, since especially the indirect impacts are often unclear and hard to quantify. The other interventions are, for example, the enhancement and protection of specific species' nesting sites, breeding programs and fishing management. The exact management interventions involved in these activities, their area-coverage and/or their (in)direct impact remain unknown. Therefore it is impossible to assess these practices using MSA values.

9.3.2 Level 2: Change in management practices

For level 2, we gathered proportional data on the interventions carried out. To assess the total direct biodiversity impacts of rewilding practices, we need to know the specific activities carried out and their corresponding area coverage. This provided the first obstacle; the absence of consistent reporting on the activities and their scale. A rough table was created with coverage data for the interventions, based on information from annual reports and the Rewilding Europe website. However, for some interventions no numbers or contradictory numbers were found. This table was sent to Rewilding Europe to check whether the numbers were correct, and whether they could provide us with more data, resulting in Table 9.1.

Out of the 4.97 million hectares, 88,130 hectares, or 1.8%, can be ascribed to the above-mentioned ecosystem-wide management practices. Again, this does not take into account the other activities (e.g. nest-protection, outreach, breeding programs) carried out by Rewilding Europe. It does show, however, that the current area influenced does not correspond to the total area as was found in level 1, and therefore highlights the importance of the level 2 analysis. Another challenge was posed by the 'connectivity' intervention. While the corridor is within the rewilding area, and the scale is known, the biodiversity impacts are possibly mainly found in the national parks connected by this corridor. Therefore it poses an issue of which area size to use; the area of the corridor (which, in this case, is not an entirely natural corridor, but a co-existence corridor), the area of the national parks (144,933 ha), or both with different impact levels?

While we roughly know the management practices and their corresponding scope in level 2, reference situations to calculate ΔMSA in level 3 were unknown. The baseline 'conventional management' does not apply here, since it really depends on where rewilding interventions are implemented. To assess the contribution of Rewilding Europe's activities to the local MSA, a baseline MSA as well as an ex-post MSA value has to be determined. To determine the baseline MSA value, the state of the area before the project interventions needs to be clear and fall within the scope of the currently existing MSA categories (e.g. 'ungrazed abandoned rangelands', 'man-made grasslands', 'lightly used natural forest', 'moderate flow deviation' and for connectivity surface area; between 0 and 10,000 km^2). The same accounts for the ex-post assessment. It has to be clear what the state of the area is, or is expected to be, after the intervention, and an MSA category in line with this has to be available. For the areas covered by Rewilding Europe, this was not available. While there was some information regarding some aspects of the land, such as abandonment, it is unclear what the state of the land is where the interventions took place. Furthermore, if assumed that these practices take place on abandoned landscapes, the time since abandoned can strongly influence the reference MSA value before rewilding actions took place (see Schipper et al., 2016).

Table 9.1
Proportional data on Rewilding Europe's area based activities

Area	Targeted scale of project (ha)	Natural grazing (ha)	River restoration (ha)	Restoring food chains (ha)*	Rewilding Forests (ha)	Connectivity (ha)	Wetland restoration and reflooding (ha)	Agreements with landowners (ha)
Lapland	3600000		X (500)	X (300)	X (70 ha)			870
Oder Delta	250000		X (50)		X (1500 ha)			1550
Velebit Mountains	220000	X (900)		X (17000)	X (400)			18300
Western Iberia	120000	X (1600)			X (1990)			3590
Central Apennines	100000					X*** (40.000 ha corridor, but connecting an area of 144933 ha		
Rhodope Mountains	250000	X (2400)		X (13700)				16100
Danube Delta	180000	X (2750)****					X (1770)	4520
Southern Carpathians	250000	X (3200)**						340
Total (ha)	4970000	10850	550	31000	3960	40000	1770	

' Agreements with hunting associations or RE taking over hunting concessions to increase populations of (mainly) deer as key species for vegetation, prey for large predators/scavengers. Often in combination with wildlife watching. ** This is the home range of a released group of (collared) bison, now formally protected by Romanian law. But only 340 ha (pre-release sites) in signed contracts. *** Dozens of apiaries and orchards fenced out with electric fences to protect them against bears. Finally resulting in almost 40,000 ha of safe corridors without human–wildlife conflicts. **** With 40,000 ha more in the pipeline, under a new project that will be approved ...' (Wouter Helmer, pers. comm., 2018).*

9.3.3 Level 3: Biodiversity impacts

The level 3 analysis, and therefore level 4 as well, again provided some obstacles, mainly relating to the lack of specific information on the baseline state of the areas in level 2, on the management practices that make up the activities mentioned in Table 9.1, and due to the limitations of the method using MSA values. For some of the activities, such as river restoration, the exact interventions are unknown. This makes it difficult to connect it to an MSA category. If dams are removed, for example, it matters greatly which dam, or how many dams are removed when calculating the MSA values. Furthermore, it is unclear if dams are removed or if other area-based interventions take place. The same was the case for rewilding forests, wetland restoration and reflooding, connectivity and the restoration of food chains. The exact interventions remained unknown and therefore these interventions could not accurately be combined with MSA categories.

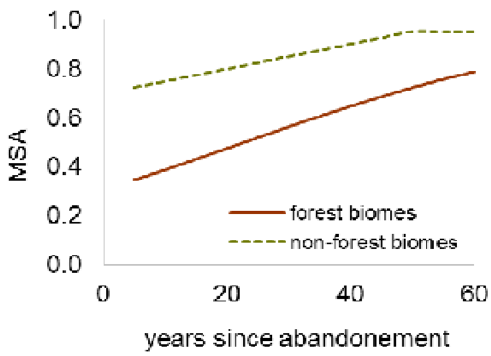
This last issue also relates to the (current) limitations of assessment using MSA values as indicators for positive impacts of ICIBs on biodiversity. There are certain activities of which the corresponding MSA value is known, but if an activity does not fall within these categories, it takes time to create a new MSA value. Therefore, we had to work with a limited data set and a limited list of management types and their complementary MSA values. In the next section, three rewilding interventions are shown that we were able to assess partially, providing a showcase of the potential of rewilding practices. The other interventions could not be assessed due to a lack of information on interventions and MSA categories. When the aspects of rewilding interventions become more clear and readily available and when more empirical studies appear on the impacts of such interventions, a next step could be to calculate MSA values specifically for rewilding interventions.

Natural grazing

For natural grazing a corresponding MSA category is available. However, some challenges still remain, such as determining the baseline MSA value for the respective category. Rewilding Europe mainly introduces natural grazing regimes in abandoned landscapes. It is unclear what the specific baseline is of the areas used for REs natural grazing interventions. Areas can (partly) recover naturally and therefore the years since abandonment influence MSA values. Schipper et al. (2016) show that for non-forest biomes, MSA values can increase from roughly 0.7 several years after agricultural land abandonment, to roughly 0.95 around 50 years after abandonment. This corresponds with the MSA values provided by Alkemade et al. (2013) on ungrazed abandoned rangelands (Table 9.2). However, it is unclear whether some of the lands covered by Rewilding Europe's activities are still (partly) in use. Therefore, we take a baseline of moderately used rangelands for these areas to be on the safe side. It is important to note that the table presented by Schipper et al. (2016) is based on limited data points for non-forest biomes. Currently, research is being conducted on a new, overall assessment. In this research, forest and non-forest biomes are combined due to the lack of data for non-forest biomes and consequently a low certainty for the relationship between non-forest biomes and MSA values (Aafke Schipper, pers. comm. 2019). The results from this unpublished analysis shows MSA values for secondary or restored vegetation in comparison to a reference situation. In this case MSA values start at roughly 0.2 immediately after abandonment, and reach around 0.6 150 years after abandonment. This shows a rather less positive story.

Rewilding Europe aims to create natural rangelands through their grazing interventions. Therefore the aim is to reach an MSA value of 1.0 in the previously abandoned areas. Since these areas by themselves can recover to an MSA value of 0.95, an additional intervention by Rewilding Europe could possibly increase this MSA value indeed to 1.0. Since we do not know the time since abandonment for these areas, we now assume an average baseline of moderately used rangelands (MSA = 0.6, Alkemade et al., 2013). This results in a best case scenario of moderately used rangelands to completely restored natural rangelands, causing an increase in MSA value of 0.4, and a worst-case scenario of a baseline of moderately used rangelands and no positive influence of REs interventions in which the system would eventually reach 0.95, so an increase of 0.35 (Schipper et al., 2016). Ascribing an MSA value of 0.35 or 0.4 to REs interventions therefore seems too optimistic. However, since Rewilding Europe works through agreements with land owners, they can ensure avoided loss. So that even if the introduction of large grazers has no net benefit, the area is still under protection through these agreements, and therefore gets the chance to recover by itself.

Figure 9.1
MSA recovery over a time period after agricultural land abandonment.



These numbers are based on MSA values from secondary vegetation derived from studies with a varying number of years since either clear-cut felling and/or land abandonment. The recovery pathways differ between forest biomes and other biomes. Biomes are derived from the IMAGE model. Figure derived from Schipper et al. (2016).

Table 9.2
MSA categories for rangelands and rangeland management, and their corresponding activities.

RE activity	MSA category	Description	MSA Globio
Natural grazing	Natural rangelands	Rangeland ecosystems determined by climatic and geographical circumstances and grazed by wildlife or domestic animals at rates similar to those of free-roaming wildlife (Alkemade et al., 2013)	1.0; natural grasslands, (Alkemade et al., 2013, Schipper et al., 2016)
	Ungrazed abandoned rangelands	Original grasslands no longer in use, lacking wildlife grazing and no forests developed (Alkemade et al., 2013)	0.7 (Alkemade et al. 2013)
	Moderately used rangelands	Rangelands with higher stocking rates: grazing has different seasonal patterns or vegetation structure is different compared with natural rangelands (Alkemade et al., 2013)	0.6; pasture, moderately to intensively used (Alkemade et al., 2013, Schipper et al., 2016)
	Intensively used rangelands	Rangelands with very high stocking rates: grazing has different seasonal patterns and vegetation structure is different compared with natural rangelands (Alkemade et al., 2013)	0.5 (Alkemade et al., 2013)
	Man-made grasslands	Rangeland with high degree of human management, including converted forests (Alkemade et al., 2013)	0.3; man-made vegetation/pasture (Alkemade et al., 2013, Schipper et al., 2016)

MSA for connectivity
For a calculation of the MSA related to increased connectivity, we take data from Alkemade et al. (2009) in which the size of natural areas is related to MSA values (Table 9.3). We have only received confirmation on the size of the central Apennines corridor and will therefore only take this corridor into account. This corridor aims to increase the connectivity between three national parks — the Abruzzo, Majella National Parks, and Sirente Velino Regional Park. The Abruzzo National Park covers 50,000 ha[43], Majella covers 74,095 ha[44] and Sirente Velino Regional park covers 56,450 ha[45], totalling 180,545 hectares. We assume that these three national parks resemble natural areas. When looking at the table in Alkemade et al. (2009), areas between 10,000 and 100,000 hectares have an MSA value of 0.9. Areas bigger than 100,000 hectares but smaller than 1 million hectares have an average MSA value of 0.95. Therefore, we take a baseline of the areas all separated, which gives us an MSA value of 0.9. If the corridor provides sufficient connection in the best-case scenario, the MSA could increase to 0.95, illustrating an MSA value increase of 0.05. In the worst-case scenario, the corridor will not add to the connectivity in the parks, leading to an MSA increase of 0.

The 40,000 hectare corridor is not added to the total, since the table provided by Alkemade et al. (2009) regards natural areas. Since the corridor is a co-existence corridor with all kinds of

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[43] http://www.parcoabruzzo.it/page.php?id=251
[44] https://www.parcomajella.it/en/park-authority/the-park/
[45] https://www.protectedplanet.net/parco-regionale-naturale-del-sirente-velino-regional-provincial-nature-park

interventions to reduce human–wildlife conflicts and increase connectivity between the parks, but not necessarily aimed at increasing the naturalness of the area itself, we take the impact on the natural parks, so between 0 and 0.05 increase in MSA value in an area covering 180,545 hectares, as a result of the connectivity intervention. Having mentioned this, it can be stated that these calculations are not conclusive. Since there is no measurement for the effectiveness of the corridor, it is an assumption that the three areas will truly be connected.

Table 9.3
The relationship between area and corresponding fraction of species assumed to meet their minimal area requirement.

Area (km2)	MSAp	SE
<1	0.3	0.15
<10	0.6	0.19
<100	0.7	0.19
<1,000	0.9	0.20
<10,000	0.95	0.20
>10,000	1.0	0.20

Extracted from Alkemade et al. (2009).

MSA for River Restoration
For the assessment of river restoration, a possibility is looking at hydrological disturbance in terms of flow deviation. Janse et al. (2015) provide MSA values for flow deviation, as well as an overview of MSA values in rivers and streams subjected to land use in the area. To calculate the baseline MSA value for this area, flow deviation and land use percentage needs to be known. For example, if the flow deviation is moderate (MSA = 0.6), and the land use 20%, then the baseline MSA for this area with these two pressures combined will be 0.9*0.6=0.54. While Rewilding Europe probably aims to get these rivers back to their natural state, and therefore an MSA value of 1, rivers undergo influence from upstream interventions. This makes it a challenge to decide on an MSA value after the interventions. Therefore research has to be conducted on the MSA of rivers after (certain) rewilding interventions. Of course, there might be more activities such as fishing bans and spawning ground restorations, which we are currently unable to assess.

9.3.4 Level 4: Implementation effectiveness

Unfortunately no systematic literature could be found on the impacts of Rewilding Europe's practices on biodiversity. This possibly has to do with the fact that RE is still a rather new initiative which aims mostly at long-term effects. Rewilding Europe does provide annual reports, which however, mainly give focus on the implemented activities, and less on the environmental impacts of these implementations. Additionally, while empirical evidence for the biodiversity benefits of trophic rewilding remains scarce, there is some evidence of improved seed dispersal and endemic tree recruitment, the suppression of invasive plants and the restoration of trophic cascades leading to regeneration of certain tree species and other indirect effects (Svenning et al., 2016).
Zu Ermgassen et al., (2018) give a brief overview of the results of rewilding management on ecosystem services, based on publications extracted from Web of Science. These studies show mixed results on the effects on ecosystem services, with some showing trade-offs between ecosystem services. Biodiversity comes up in two of the four studies mentioned: one of which mentiones a negative impact (Cordingley et al., 2016), the other a positive impact (Navarro and Pereira, 2015). This last study conducted a qualitative assessment on the effects of rewilding on

ecosystem services, and concluded that rewilding is the most effective strategy for providing habitat for biodiversity compared to extensive agriculture areas, plantation forests and intensive agriculture areas. Navarro and Pereira (2015) also provide an assessment of the benefits of rewilding for biodiversity specifically. In a review of 23 studies, they found a positive response of species to either a decrease of human pressures, or to restoration following land abandonment. The results provide an expansion of an earlier study by Russo (2006) examining the effects of land abandonment on animal species in Europe. Navarro and Pereira (2015) identified 110 species as benefiting from land abandonment and rewilding, while 101 species were negatively affected by it. 13 of these species were reported as being both negatively as well as positively affected, depending on the study and study region.

Benayas et al. (2009) conducted a meta-analysis of 89 restoration assessments and the effects on, among others, biodiversity. While this study focused on a wide array of restoration strategies, so not just rewilding activities, a limited amount of studies covered strategies used by Rewilding Europe (reintroduction of herbivores or carnivores, 3 studies; passive restoration — halting of degrading action only, 13 studies). The results indicated that ecological restoration increased provision of biodiversity by 44%. However, the values in restored systems remained lower than those of intact systems. This could give an indication that (certain) rewilding practices could contribute to biodiversity, but might not lead to an entirely intact reference ecosystem.

There is an urgent need for empirical evidence of rewilding impacts on specifically biodiversity. More quantified impact assessments are necessary on the changes in biodiversity after rewilding interventions. Pettorelli et al. (2018) provide an overview of examples of targets that might be implemented by rewilding initiatives and how to measure the actual outcomes for these targets.

9.4 Discussion and conclusion

We were able to assess the scope, practices and partly the impact of Rewilding Europe's practices. With a coverage of 4.97 million hectares that continues to grow, Rewilding Europe has a big potential to impact biodiversity in Europe. The practices carried out focus on the restoration of nature and show that they could have the potential to increase biodiversity. Since limited studies exist on the actual impacts of rewilding, the results of level 3 provides more of a preliminary estimation that still has to be fine-tuned. Nevertheless, it shows how MSA increase can be assessed through the framework.

While this chapter shows interesting outcomes for the different levels of the assessment framework, due to a lack of full data on the scale, management practices, baseline states, impacts and some limitations in using the GLOBIO model method which uses MSA as a single indicator, a full MSA calculation for the whole Rewilding Europe area was impossible. With some assumptions a level 3 analysis could partly be done, the results do come with a high degree of uncertainty. To solve these issues, several steps have to be taken. Reporting has to become consistent, sufficient and available, more impact studies have to be undertaken on the impacts of rewilding efforts upon which more MSA categories can be created to support robust and coherent impact assessments for rewilding activities throughout the world.

Next to these data and methodological limitations, the question that arises is, how much rewilding contributes to abandoned landscapes that 'rewild' by themselves? What is the value added of rewilding practices to such landscapes? As shown in Schipper et al. (2016), non-forest abandoned landscapes seem to be able to almost entirely recover by themselves. For forest abandoned landscapes this remains uncertain due to an absence of data past 60 years since abandonment. The question is, what the added value is of certain rewilding interventions compared to a hands-off policy? This will probably depend highly on the baseline state of an area, as well as the pressures that an area undergoes. Important to keep in mind as well, is that under rewilding management, there is always the additional benefit of avoided loss compared to unmanaged abandoned landscapes. Rewilding interventions could possibly accelerate the recovery process or enhance the state of the area, but this again depends on the baseline state.

One rather controversial issue regarding rewilding is the naturalness of Rewilding Europe's areas. Even though this issue presents a sidetrack from this case study, it is important to mention briefly. There are two main hypotheses regarding the primaeval forests of lowland Europe; the wood–pasture hypothesis as supported by Rewilding Europe, and the high-forest hypothesis. Rewilding Europe bases their practices on the ideas of Vera (2000) that large herbivores dominated and maintained open landscapes, wood–pasture landscapes. These ideas are highly contested and receive criticism from other scholars. Vera based his ideas on pollen data. Since then, this hypothesis has been rejected based on pollen data analysis by Mitchell (2005), concluding that there was no indication of open forest canopies at the researched areas before human impact[46]. Since thinking about the proper reference system to guide management practices is highly relevant, this discussion might lead to some different assumptions regarding when a rewilding area reaches its natural state, or an MSA value of 1.0. Therefore, it might be beneficial to know the state of an area before rewilding interventions, and after. In that way an increase in MSA value can directly be attributed to rewilding practices, instead of inferring MSA value by assuming the area reaches the 'original' state of MSA = 1.0.

Since there are many uncertainties regarding rewilding practices, it is important to properly assess the practices and impacts involved. Reporting on the scale, impacts, and state of rewilding projects needs to increase, as well as impact studies of the several management interventions used in rewilding projects. Furthermore, an update and upscaling of MSA values would support further impact assessments of overall rewilding initiatives, but to calculate these MSA values, again, more empirical research is needed.

The concept of rewilding is gaining momentum as a promising restoration strategy since it has the potential to not only conserve natural heritage, but also to generate several co-benefits (zu Ermgassen et al., 2018) in line with Kunming-Montreal Target 11 to restore essential ecosystems and with several of the broader Sustainable Development Goals. Furthermore, the European Rewilding Network is expanding, and new initiatives pop up all around the world. By systematically keeping track of these initiatives and their impacts, their contribution to several of the new targets for the post-2020 biodiversity plan could be assessed. A recent article by Torres et al. (2018) provides a good first step in this direction. If the rewilding values presented there could somehow be connected to MSA values, this would provide an opportunity to assess the rewilding impacts on biodiversity as well.

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[46] For more information, read for example Birks, H.J.B. "Mind the gap: how open were European primaeval forests?." Trends in ecology & evolution 20.4 (2005): 154-156; Samojlik, T., & Kuijper, I. (2013). Grazed wood-pasture versus browsed high forests: Impact of ungulates on forest landscapes from the perspective of the Białowieża Primeval Forest. In Trees, Forested Landscapes and Grazing Animals (pp. 159-178).

10 Community Conservation

Since the 1980s the potential of indigenous and community conserved areas has increasingly been recognised as an influential approach in biodiversity conservation. Although indigenous people make up only 5% of the global population (World Bank, 2019), they have ownership or management rights over up to a quarter of the global land area (Garnett et al., 2018). When including local communities, both of these groups' customary lands together are estimated to cover at least 50% of the global land area (Oxfam, 2016). Furthermore, their territories coincide with a substantial amount of the world's terrestrial biodiversity (Sobrevilla, 2008; Gorenflo et al., 2012; Garnett et al., 2019), highlighting the valuable role they play in bending the curve of biodiversity loss.

Building on the historical relationships between Indigenous People and Local Communities (IPLCs) and their lands, this type of conservation combines the fulfilment of local people's needs and the conservation of nature. From generation to generation, IPLCs have built up a holistic knowledge and understanding of their lands, often leading to sustainable management. Against a set of criteria, many of these areas are now referred to as Indigenous and Community Conserved Areas, or ICCAs (for a schematic overview see Figure 10.1). Such ICCAs are defined as 'natural and/or modified ecosystems, containing significant biodiversity values, ecological benefits and cultural values, voluntarily conserved by indigenous peoples and local communities, through customary laws or other effective means' and encompass three essential characteristics: (1) an identifiable people or community are related to them, (2) this local people or community hold the power in decision making and implementation regarding governance and management, and (3) the local management decisions lead to conservation of habitats, species, genetic diversity, ecological functions and associated cultural values (Kothari et al., 2012). The term presents a convenient way to cover the vast diversity of terms and practices used throughout the world.

As shown by the third criteria for ICCAs, they do not per definition, have conservation of biodiversity as the primary objective of the community. This highlights a crucial difference with a protected area which is defined by IUCN as 'a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values' (Dudley, 2008). ICCAs can fulfil many objectives of which conservation can occasionally just be an outcome. On account of the main objective of this study, for this case study we will focus on the biodiversity impact of ICCAs alone. If properly managed, these ICCAs can for example, reduce deforestation (Porter-Bolland et al., 2011; Ding et al., 2016), tropical forest fires (Nelson and Chomitz, 2011), help in conserving critical and threatened species (Kothari et al., 2012; Bhagwat et al., 2006), provide corridors (Kothari et al., 2012; Bhagwat et al., 2006).

Figure 10.1

Indigenous and Community Conserved Areas

As a concept, indigenous and community-based conservation developed as a response to the failure of the management practice of exclusionary, or fortress, conservation (Berkes, 2007). The consolidation of the concept is further supported by a growing academic consensus on the contributions of local people to conservation, and the importance of involving them. At the same time, a global shift in the perspective on governance created a pivotal change towards a multi-level or bottom-up governance approach to protected areas (Tole, 2010; Rodríguez-izquierdo et al., 2010). Supporting this, IUCN came up with governance, shared governance, private governance and governance by indigenous peoples and local communities (Dudley, 2008). While some ICCAs comply with the definition of these protected areas, the land management of indigenous and local communities comes in diverse ways. Later, such 'alternative' conservation methods entered into both international law through for example Agenda 21 and the Convention on Biological Diversity's Kunming-Montreal Target 11, which requests that governments conserve nature through not only protected areas, but also Other Effective area-based Conservation Measures (OECMs). Noting that OECMs can be very diverse in terms of governance actors and practices, many such Indigenous and Community Conserved Areas (ICCAs) can be characterised as OECMs. Since only about 21% of indigenous lands are within protected areas (Garnett et al., 2018), now, through the introduction of the term OECMs, the remaining indigenous and community areas can too play an official role in nature conservation.

Despite the increasing attention for OECMs, ICCAs continue to face serious threats to their existence. Kothari et al. (2012) name several of these threats; 'damaging 'development' and commercial projects, imposition of inappropriate land uses, pollution and climate change, demographic and cultural changes, and economic or political inequities' and note that due to the lack of recognition of ICCAs, these threats are greatly exacerbated. While international recognition of ICCAs is increasing, the recognition and documentation of OECMs such as ICCAs by governments, remain poor and it was only recently at the Convention on Biological Diversity's (CBD) 14th Conference of the Parties that, after almost a decade, an official definition of OECMs was adopted[47]. This has led to the absence of OECMs in databases such as the World Database on

[47] UNEP/CBD/COP/14/L.19

Protected Areas (WDPA) and the poor documentation of ICCAs in general, with only 1377 ICCAs reported on in the WDPA in 2019. Often, this is also the consequence of a dependency on states to report on conservation areas, and many ICCAs not being officially recognised (UNEP-WCMC, IUCN and NGS 2018). Nevertheless, several non-state initiatives have been created over the years that have increased the visibility of ICCAs such as the Equator Initiative, LandMark and the ICCA Registry. Simultaneously, while state recognition of ICCAs gradually rises (Alden Wily, 2018), a recent note by the CBD secretariat states that recognition and reporting of ICCAs is essential to reach international biodiversity targets[48]. Even though much still remains unknown, this case study will show the potential of both officially state-recognised as well as non-recognised ICCAs worldwide and their contribution to nature conservation. Since the ICCA registry is the only specific registry of ICCAs worldwide, this database will mainly be used for this case study.

10.1 Motives, goals and targets

The **motives** behind ICCAs differ between communities, from securing land tenure and sustainable livelihoods to ethical reasons and concerns for the loss of wildlife (Kothari et al., 2012). Since IPLCs can differ greatly in interests and concerns, so can their motives and practices (Table 10.1). There are several types of practices of ICCAs (Table 10.1), encouraged by various motivations which are at play. The CBD lists the main motivation as ‘life itself, including all that goes with it: survival, livelihoods, culture and identity’. However, more specific motivations can be recognised as well, such as: securing land tenure, securing sustainable provision of resources for livelihoods, concerns for local wildlife, to sustain beneficial ecosystem services, maintaining links in the land or seascape, sustaining religious, identity or culture needs, security against emergencies, generation of revenues (Kothari et al., 2012).

Consequently the **goals** of ICCAs can be very diverse as well. However the overarching goal of IPLCs behind the general motivation of ‘life itself, including all that goes with it: survival, livelihoods, culture and identity.’, seems to be to sustain and protect this life and lifestyle.

Few ICCAs set out specific **targets**. Nevertheless, their activities do contribute to several global targets. Concerning biodiversity, the closest contributions are made to the Kunming-Montreal Targets of the CBD of which specifically Target 11 (the increase of protected areas up to 17% on land and 10% in the ocean) is often mentioned in relation to ICCAs (UNEP-WCMC, IUCN and NGS 2018; Lopoukhine and Ferreira de Souza Dias, 2012; Woodley et al., 2012) and to Sustainable Development Goal 15 (protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, halt and reverse land degradation and halt biodiversity loss). Furthermore, specific ICCAs can contribute to a diverse range of other international targets as well.

10.2 Theory of change: from input to impact

Input by indigenous and community conservation is not so much aimed solely at conserving biodiversity, as in most other case studies, but comes from an intent to achieve a state of

[48] UNEP/CBD/COP/14/INF/25

integrated wellbeing, which entails being part of a well-functioning ecosystem (Alcorn, 2010; Kothari et al., 2012). The output is usually a result of generations of trial and error and covers the knowledge and rules gathered throughout this history. While indigenous and traditional ICCAs are mainly based on this traditional knowledge and accompanying rules system, modern ICCAs might have strategic plans to obtain their goals, accompanied by stricter rules and regulations. The outcomes can be seen as improved well-being and/or positive biodiversity impacts.

10.3 Past performance: assessing outcome and impact

ICCAs provide a special case study in this report. While community conservation is increasingly getting attention due to their potential positive impact, they are far behind community forest management (Chapter 7) in terms of recognition and research. Nevertheless, they reflect an ever growing movement, which is why we aim to show their potential impact as far as we can go through the different levels of the framework.

10.3.1 Level 1: Global reach

Different estimates exist for the land used and owned by IPLCs. Alden Wily et al. (2011), estimates the 'community owned commons' at 6.8 billion hectares, or over 50% of global land area[49]. These community owned commons are defined by Alden Wily et al. (2011), as 'lands which rural communities possess and use collectively in accordance with community-derived norms', also referring to indigenous peoples and community lands. It is important to note, however, that ownership of the greater proportion of these lands, is not necessarily with the local communities, but often vested in the state. A more recent estimate focuses on the area owned by indigenous peoples while leaving out the notion of local communities (Garnett et al., 2018).

The authors recognise that in some places the terms Indigenous and local community are synonymous, but focus on indigenous peoples because of the availability of maps of their lands. Their results show that at least 3.79 billion hectares is owned or managed by Indigenous Peoples, or 28.1% of total land area. Of this area, 20.7% is within protected areas, covering at least 40% of the global protected areas and thereby making up the highest proportion of land type within protected areas. The authors only found data on Indigenous lands in 87 of 235 countries and administratively independent entities, indicating that these numbers could be an underestimation of the total area owned or managed by indigenous peoples. Furthermore, to these numbers data should be added on the extent of non-indigenous local communities, however, no comprehensive global data on this number is currently available.

Another source of data is the Landmark website[50] covering both the lands of communities and indigenous peoples. Currently, the Landmark maps cover 12.4% of the world's land based on boundary data. When looking at the background data (Dubertret and Alden Wily, 2017) extracted

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[49] Garnett et al. 2018 work with a total land area of 13.52 billion ha, while Wily 2011 works with a total land area of 13.066 billion ha. Both exclude Antarctica (and uninhabited islands in the Southern Ocean Garnett et al. 2018).

[50] www.landmarkmap.org

from literature, roughly 5 billion hectares (38%)[51] is held or used by Indigenous Peoples and Local Communities. While the percentage data is more comprehensive than the boundary data, both still have gaps for many countries and data is therefore incomplete (Katie Reytar et al., 2018).

This leaves us with an idea of the extent of indigenous and community lands, nevertheless, not all of these lands fulfil conservation objectives. The extent of lands conserved by IPLCs remains unknown. This information is needed to get an idea of how much IPLCs are contributing to the conservation of areas worldwide. Kothari et al. (2012) gives an estimate of the scope of ICCAs within 21 countries. When adding these numbers (taking the lowest number in cases where a range is given), **one arrives at an estimate of 294 million hectares covered by possible ICCAs**. The numbers in this study are only estimates, and fit either the ICCA definition or contribute to conservation in other ways. The ICCA registry is currently making progress towards providing more comprehensive information on the amount and extent of ICCAs. However, **in 2018, the ICCAs registered in this registry covered 27.9 million hectares** (Heather Birmingham, pers. comm., 2018)[52], roughly 0.2% of the world's land area. These ICCAs cover both marine and terrestrial areas, but nevertheless their coverage is significantly less than the numbers mentioned by Alden Wily et al. (2011), Garnett et al. (2018), the Landmark website and Kothari et al. (2012). So while the commons under registered ICCAs cover 27.9 million hectares, this is highly likely an underestimation of the total global ICCA coverage.

As you can see, the level 1 analysis for this case study already provides some challenges relating to an absence of complete data on the amount and coverage of ICCAs. While some information can be extracted from other sources, detailed and comprehensive data on the extent of ICCAs is not readily available to date. This is probably due to the relatively recent consolidation of the concept, the lack of recognition of IPLCs, and the risks recognition brings to IPLCs. Therefore, numbers suggested here might be an underrepresentation of the total area covered by ICCAs. Furthermore, the subject of research differs between studies. While some studies report on indigenous peoples in general, either globally or locally, other studies report specifically on ICCAs.

10.3.2 Level 2: Change in management practices

The level 2 analysis should cover the management practices of ICCAs and their corresponding areas. Several different types of ICCAs and their related practices have been identified by Kothari et al. (2012) (Table 10.1). Such practices have different outcomes regarding biodiversity conservation.

[51] The referenced total land area is 13.26 billion hectares

[52] This data is non-spatial and should therefore be treated with caution since overlap between different ICCAs and the correctness of provided units cannot be checked.

Table 10.1
Different types and practices of ICCAs as identified by Kothari et al. (2012).

Type	Practice
1. Indigenous people's territories	Managed areas as part of the communities history and lifestyle, regarding sustainable use, cultural values and conservation objectives
2. Nomadic territories	Territories used by mobile or nomadic communities, managing resources through unofficial regulations.
3. Sacred natural sites	Sites that are often left untouched because of their sacredness
4. Resource catchment areas	Areas that are essential to the livelihoods of communities, or that provide key ecosystem services. These areas are managed in a sustainable way.
5. Ecosystem optimisation territories	Territories conserved to support the productivity of related ecosystems.
6. Commercial areas	Sustainably managed areas for support of commercial benefits such as ecotourism.
7. Critical plant and animal habitats	The protection of critical areas for wild plants and animals such as nesting or roosting sites.
8. Mosaic landscapes	Landscapes with mosaics of natural and agricultural systems that contain important biodiversity and cultural value and that are managed by rural or mixed rural–urban communities.
9. Urban rural space	Urban green spaces with relatively high biodiversity values.

Similarly, the baseline of the land managed by IPLCs can cause differences in the effects of the practices in ICCAs. For example, a sacred-site ICCA in the Amazon protects the local biodiversity of an area without any, or with limited human impact, while an urban green space can support sufficient biodiversity to be an ICCA, but can still be under high negative influence of the environment.

35 case studies are currently presented on the ICCA registry website wherein all these case studies provide different types of information. Information on general use of an area (protected, sustainable use, resource catchment etc.) is often given in these case studies. If this information becomes a bit more detailed, such case studies can be linked to the different ICCA types mentioned in Table 10.1. Nevertheless, information on the extent of the practices in these ICCAs remain absent, which makes a full level 2 analysis impossible. Furthermore, establishing a baseline for ICCA management can be difficult since many ICCAs have been around for centuries, or even thousands of years.

The absence of this data shows the presence of a significant data gap, which additionally illustrates the sensitivity of this case study. Data on ICCAs is generally very sensitive, not all ICCAs want to be registered and/or recognised since some forms of legal recognition can be inappropriate and can lead to a variety of adverse effects such as legal uncertainty, overruling of customary rules, the undermining of customary laws, imposed procedures and protocols, and short-term instead of long-term land management decisions (Kothari et al., 2012; Stevens et al., 2016). This is also why the ICCA registry is an offline database which does not allow access with the exception of a few case studies.

10.3.3 Level 3: Biodiversity impacts

Due to the lack of a global meta-analysis on the impact ICCAs have on local species richness or abundance, no generalisable conclusions could be made on the contributions of ICCAs to MSA. The available literature does provide some insights in the contributions of community conservation practices to local biodiversity. To provide a full overview of the contributions of ICCAs to biodiversity, however, a thorough systematic literature review would have to be conducted, which does not fall under the scope of this project.

The IPBES Global Assessment (Chapter 2) mentions several ways in which IPLC contribute to nature; (1) the human-nature co-production of high diversity cultural landscapes which can host rich open habitats and high landscape heterogeneity, (2) the development of species-rich natural ecosystems for wild species which can even develop into local hotspots for biodiversity, (3) creation of ecosystems with wild and domestic species, or agroecosystems, (4) selection and domestication of ecosystems contributing to agro-diversity, (5) the enhancement of natural resilience through traditional management practices, (6) increasing net primary biomass production, (7) protecting and sustaining high conservation value ecosystems from external users. **Also, around half of the lands of indigenous peoples is primary vegetation. Two thirds of this is classified as natural (Garnett et al., 2018), compared to 39% (IPBES, 2019) and 44% (Garnett et al., 2018), respectively, of other lands**.

While several studies indicate that IPLCs contribute to nature conservation in several ways, such as through reducing deforestation (Bray et al., 2008; Armenteras, Rodríguez, and Retana, 2009; Shahabuddin and Rao, 2010; Porter-Bolland et al., 2012) and conservation and protection of wildlife habitat (Bhagwat et al., 2006; Kothari et al., 2012), not enough evidence is currently available to draw generalisable conclusions on the quantitative contributions to MSA values of IPLC, let alone ICCAs, and more quantitative studies to show causal relationships are needed.

10.3.4 Level 4: Implementation effectiveness

ICCAs were identified as a key case study to represent ICIBs within the envelope of community conservation. However, in measuring its impact this study encountered some challenges which were unable to be redressed. While this study recognises the indisputable contribution that ICCAs make in forest restoration and conservation, albeit with indirect impacts, ICCAs are still at a rather less-studied nascent stage to make conclusive impact assessments of biodiversity. Some of the key reasons for this are listed below and further discussed in the following 'Discussion and Conclusion' section. Firstly, ICCAs are considered a movement as opposed to an institutional setup which makes it difficult to categorise as a typical ICIB in and of itself. Secondly, as ICCAs are a type of community conservation, there are no clear and established boundaries distinguishing ICCA conservation and restoration activities from other forms of community conservation activities. Thirdly, this study highlights the role of ICCAs as critical for biodiversity conservation, yet because of the nascent nature of ICCAs there is no clear baseline to define the precise impact it has had on biodiversity conservation. Finally, as ICCAs are part of other forms of community conservation, it has played multiple roles not only in conservation but rewilding areas, restoring biodiversity, and preserving pristine biodiversity habitats. This adds in several unaccounted variables that in measuring the impact of ICCAs on biodiversity conservation doesn't provide a clear picture of biodiversity conservation with MSA values as a sole measure.

10.4 Discussion and conclusion

This chapter has shown how ICCAs can be assessed according to the levels of the assessment framework. ICCAs provide an interesting topic for assessment since they are more of a movement rather than an institutionalised ICIB, and since they are increasingly getting attention as an important part of nature conservation. The results of the assessment reflect the non-institutionalised character of ICCAs, since quite some information remains unknown. However, the available information to date does provide the opportunity to make an estimation on the outcome of level 1 of the assessment framework and provide insights in level 2.

Regarding level 1, no clear number exists on the coverage of ICCAs. While the ICCA registry is on its way and has registered ICCAs in total covering 27.9 million hectares (in 2018), this seems to be only a small fraction of the ICCAs out there. It is significantly less than the numbers mentioned by Alden Wily et al. (2011), Garnett et al. (2018), the Landmark website and Kothari et al. (2012) for IPLCs. Some information on ICCAs can also be found in the World Database on Protected Areas (WDPA). We have looked at the overlap between the WDPA and the ICCA registry. Out of the 35 case studies presented in the registry, 22 (63%) were not found in the WDPA, illustrating that reporting of ICCAs by governments is minimal in the WDPA. Moreover, ICCAs often do not adhere to the PA criteria but are rather OECMs, or both. The first option also presents other issues regarding community conserved areas. Whereas they have the potential to fill a large part of the conservation gap, ICCAs still suffer from a lack of legal recognition in many countries (Kothari et al., 2012; Stevens et al., 2016; Arjjumend et al., 2017). Such a lack of recognition can have several detrimental effects for both the IPLCs involved as well as for biodiversity, and undermines promising conservation opportunities. Steven et al. (2016) mention, for example, that the absence of, or insufficient legal recognition can lead to a 'tragedy of the commons'' situation, wherein incentives for community members to use areas according to customary rights, and systems to defend the area against external users are lost. Fortunately, recognition of indigenous and community lands is increasing (Kothari et al., 2012; Alden Wily, 2018; Arjjumend et al., 2017) which could potentially also lead to better registration of and more information on ICCAs and their practices and contributions to nature conservation.

Important to note is that the registration, recognition and visibility of more ICCAs should only be done with free, prior and informed consent, due to the sensitivity of the IPLCs and their way of living (Kothari et al., 2012). Therefore, having access to a comprehensive database provides a challenge. The ICCA registry is furthermore an offline database, which means that they do not release detailed data, except for the few case studies they have on the website. This is not necessarily something you want to encounter when assessing the extent of ICCAs, however, since there are many reasons for these communities to limit access to what they do and where they do it. Nevertheless, if general, global data can be made available in the future, it provides a good baseline for assessing the impact of ICCAs.

For Level 2, Kothari et al. (2012) provide valuable insights in what kind of practices take place in ICCAs. It shows that the range of activities within ICCAs is large; from modern practices to ancient practices or even leaving an area alone for spiritual reasons. It shows that in all of these cases ICCAs have the potential to contribute to biodiversity gain. Possibly due to the sensitivity of ICCAs in terms of politics, little is known about the coverage of these different practices. If this would become available through the ICCA registry for example, without details on where or which community, this could greatly help in assessing the impact of ICCAs compared to conventional practices. This could then also be linked to MSA values if corresponding categories are available. The lack of information in both MSA and the activities within ICCAs is largely due to a lack of research on ICCAs.

While research has been conducted on community forestry (see Chapter 7), research on which other types of ICCAs exist, and what their management practices and biodiversity impacts are, remains minimal. Again, as for level 1, if more research on this topic would be conducted, this could provide valuable information for level 2 of the analysis and for the creation of MSA values. Another point to keep in mind is that in many cases of IPLC conservation, the baseline for these areas often seem to be pristine areas, indicating a loss in biodiversity due to IPLC conservation when applying this method. However, IPLC conservation could rather be seen as avoided loss, or as an alternative to conventional logging or agricultural practices. This means that community conservation could provide an increase in MSA values when compared to more conventional practices, or avoid loss of more biodiversity, while keeping the natural system relatively intact. In other cases ICCAs directly can contribute to biodiversity by, for example, rewilding a degraded area. This case-study has shown that ICCAs have a large potential to contribute to global biodiversity. This potential of ICCAs to fill important conservation gaps is fortunately increasingly getting attention. For example through the recent uptake of OECMs in the WDPA, and the possibility for OECMs and ICCAs to register themselves (UNEP-WCMC, IUCN and NGS, 2018). This increasing recognition and reporting can eventually lead to better assessment of their contribution to biodiversity governance, and thereby filling important conservation gaps.

11 Conclusions and Reflections

This report aims to quantify the contribution of international cooperative initiatives (ICIB) towards the conservation and sustainable use of biodiversity; to further develop a methodology to assess the biodiversity impact of non-state action; and to review methodological challenges taken from this assessment. This type of assessment and information is relevant for the further development and evaluation of efforts under the non-state action agenda for biodiversity of the Kunming-Montreal Global Biodiversity Framework of the CBD. Eight cases were analysed: **(1) Community Conservation, (2) Agriculture Voluntary Sustainability Standards in agro-commodities (AVSS), (3) Organic Agricultural certification, (4) Rewilding Europe, (5) Community Forest Management (CFM), (6) Forest certification, (7) ICLEI, and (8) Integrated Landscape Management (ILM)**.

These cases were described through an analysis of their motives, goals and targets, and their theory of change, followed by the core of the assessment: a cautious analysis of the potential impact of ICIBs. This impact assessment was conducted based on an assessment framework that we developed building on earlier work by Arts et al. (2017). The method is described in Chapter 2.

The assessment framework consists of four levels: (1) an assessment of the global reach and/or number of initiatives of sustainable managed or conserved areas by ICIBs, (2) the identification of a change in management practices by a specific ICIB compared to the pre-intervention situation, and (3) an assessment of the potential biodiversity impact per management practice under the previous level (expressed as Mean Species Abundance or MSA), as well as an estimation of the total MSA-increase per hectare for the area as a whole (compared to conventional management practices). Level (4), finally, addresses the implementation effectiveness of an ICIB. At this level, it is estimated how many of ICIB's initiatives seem to have been truly adopted on the ground and may be expected to truly realise the positive biodiversity impacts indeed. For most cases in this report, however, level 4 could not be estimated, which we address in 11.2 below.

This final chapter will summarise the findings of the individual case studies, followed by a synthesis of these findings addressing the first research question (Section 11.1), data challenges we faced (Section 11.2) and methodological reflections on our approach (Section 11.3). The latter relates to our second research question and will draw lessons from our methodological approach for assessing the impacts and contributions of non-state actors for the Action Agenda for Nature and People. We discuss some broader issues about this study in Section 11.4, and present a conclusion in Section 11.5.

11.1 Findings from case studies

Our case studies can be positioned in the Outcome and Impact part of the I-O-O-I scheme (see Figure 2.1, Chapter 2). This scheme presents a dynamic and basic concept for evaluating and assessing impacts in many different disciplines and can be used for different types of activities. Outcome and Impacts can, for example, be assessed at the level of financing and networking, in which an outcome can be the number of members being active in an initiative, and impact the long-term biodiversity effects of this initiative on the ground. This impact can then be monitored and will provide a basis for new input. In this report, the outcome and impact relate directly to area-based activities and the on-the-ground biodiversity changes. All the initiatives have gone through the stage of becoming a concrete idea (input), followed by being created, institutionalised or formalised (output), upon which we address the outcome and impact. The outcomes of the case studies clearly relate to level 1 and 2 of our impact assessment (the total area / number of initiatives and change in management practices), whereas the impact relates to level 3 (the impact calculations with MSA) and level 4 (implementation effectiveness) (see Chapter 2, Figure 2.1).

Several observations can be made regarding the I-O-O-I scheme. Firstly, having monitoring systems are part of the output of an initiative, and while many other outputs can be identified, the absence of such a monitoring system in several of the case studies, as well as the absence of impact assessments, was a major barrier for impact analysis. Additionally, in some cases the I-O-O-I scheme can be seen as working ‘in the opposite direction’. Community conservation, for example, can be a gradual process of thousands of years that have led to current practices/impacts. Recently, it has been identified that these practices are beneficial to biodiversity conservation. So the impact is known, but no information on the scale (outcome) and the organisation (output) of these communities actually exists.

Summing up all of the case study outcomes on level 1, we see that the ICIBs assessed in this report cover 100s of millions of hectares, although uncertainties and bandwidths per ICIB are high, or even unknown (Table 11-1). These numbers exclude the ICLEI initiative, since it was not possible to provide a coverage of the initiative in terms of hectares. Important to take into account is that while some ICIBs carry out activities in the whole area covered, others only act in patches of this total area. Furthermore, the coverage of the assessed initiatives overlap. For example, in the case of CFM and forest certification, around 5 million ha do overlap, and around 20 million ha for CFM and Community Conservation (which is negligible on a 100 million hectares scale). For the other initiatives, we unfortunately do not know the degree of overlap.

All cases do show changes in management practices (level 2), compared to the pre-intervention situation, but these can be of a very different nature (compare for example the protection of traditional sacred sites in tropical forests versus building green infrastructures in hypermodern global cities). Most of these interventions are estimated to have a positive biodiversity impact (level 3). This is the case wherein projects are actually implemented and are assumed to become effective over time indeed. For as far as we know from two case studies — forest certification and community forest management — implementation effectiveness (level 4) amounts to about 50 to 75% of projects, although biodiversity impacts realised may still be small.

We have been able to gather relevant information regarding the outcomes and impacts of ICIBs on biodiversity, such as their global reach, their changes in management practices that positively influence biodiversity, an indication of the increase in MSA values (per management practice in various cases and over the whole area in one case) and an estimate of the implementation effectiveness (in two cases). Due to several limitations which we discuss below, we unfortunately could not complete an analysis of level 3 and 4 in all chapters/cases. Nevertheless, CFM and forest certification demonstrate good examples of how this could be achieved when more resources and data become available for all ICIB cases.

Table 11.1
Overview of the assessment of the case studies.

Case studies	Level 1: Global reach (ha)	Level 2: Change in management practices	Level 3: Biodiversity impact in ΔMSA	Level 4: Implementation effectiveness
Agricultural Voluntary Sustainability Standards (AVSS)	Somewhere between 12 and 16 million ha	Yes (e.g. water, soil, habitat protection)	N.A. (but studies reveal some positive impacts)	N.A.
Organic Agriculture Certification (OAC)	About 60 million ha	Yes (e.g. water, soil, habitat protection)	N.A. (but studies report higher species richness / abundance)	N.A.
Community Forest Management (CFM)	Somewhere between 500 and 700 million ha	Yes (e.g. replanting, enrichment)	On average 0.2 and 0.5 ΔMSA (depending on intervention)	About three quarters of CFM initiatives produce (some) positive biodiversity impact
Forest certification (FC)	About 420 million ha	Yes (e.g. reduced impact logging, protected forest plots)	Avoids biodiversity loss with about 30% compared to conventional logging (ha*MSA)	About half of FC initiatives produce (some) positive biodiversity impact
Integrated Landscape Management (ILM)	About 360 million ha	Yes (e.g. investments in natural resource management, new reserves)	About 0.2 ΔMSA (based on one scenario study)	Nearly three quarters of IFM initiatives produce (some) positive biodiversity impact
ICLEI	N.A. (but membership of nearly 900 cities around the world)	Yes (e.g. green urban infrastructure, nature-inclusive city development)	N.A.	N.A.
Rewilding Europe (RE)	About 5 million ha	Yes (e.g. natural grazing, wetland restoration)	About 0.05 and 0.35 ΔMSA (depending on intervention)	N.A.
Community Conservation (CC)	Somewhere between 30 and 300 million ha	Yes (e.g. protection of sacred and catchment sites)	N.A. (but high proportion of ‘natural sites’ on indigenous lands)	N.A.

‘N.A.’ indicates that data is not available. All figures are estimates with (mostly unknown) uncertainties

11.2 Data challenges

To answer the second research question 'What methodological lessons can be learned from this analysis for the assessment of the impacts of non-state actors and voluntary commitments?', we will summarise the data and methodological challenges and lessons learned from the case studies here. More in-depth information for each case study can be found in the corresponding chapters.

Due to differences between the initiatives in terms of coverage, approach and data availability, not all case studies produced complete results. Forest Certification and CFM provided sufficient information, both in terms of data availability and the methodology, to be assessed at all 4 levels (although various gaps and uncertainties did still pop up in these two assessments). A complete analysis is especially true for Forest Certification, where the overall biodiversity gains from ICIBs could also be expressed in terms of ha*MSA. The other case studies were however subject to data gaps, methodological limitations, or both, so that not all 4 levels could be addressed. In this section we will discuss the limitations with respect to data.

In terms of level 1, the outcome data, we were able to assess the coverage of all initiatives except for ICLEI, providing a good overview of the spatial scope of potential biodiversity impact the initiatives may have. The data gathered per initiative was, however, very mixed. Whereas for some of the case studies level 1 data on global reach and coverage of the ICIBs could be found either through databases or research articles, other case studies already provided challenges at this level. The latter case studies have either recently been initiated, or have only recently been consolidated, which limits the amount of (outcome) data yet available. Especially for some of the case studies focusing on movements with less institutionalisation like Integrated Landscape Management and Community Conservation, it turned out to be very difficult to find comprehensive data on the extent of their activities, let alone the spatial coverage of management practices carried out.

Due to the relatively recent consolidation of the concept of community conservation, for example, and due to political reasons, databases are incomplete and do not provide a complete representation of the scale of this ICIB. Since this is already the case for level 1, a level 2 analysis is even more difficult. Whereas some information is available concerning the practices carried out in community conserved areas, no hard numbers can be extracted. For ICLEI, level 1 already provided challenges in terms of area coverage and assessing the effectiveness of biodiversity strategies. While the amount of cities with biodiversity strategies could be found in the ICLEI database, no assessment of the coverage of the biodiversity-positive activities was available, probably due to the different nature of the changes in management practices in cities compared to initiatives in rural areas. Rewilding Europe posed its own challenges, as no database with a clear overview of the total area on which Rewilding Europe carries out certain practices was available. Some information was available on Rewilding Europe's website, however, and we received more detailed and comprehensive data through personal contact with Rewilding Europe. This data did provide some uncertainties, since only part of the area mentioned to be targeted by Rewilding Europe actually undergoes management changes compared to the pre-intervention situation.

While for most cases information was available on 'biodiversity-positive' management practices by the initiatives, level 2 could often not be fully examined, similarly due to a lack of reporting on coverage. No coherent data sets are currently available showing the (ratio of) management practices for initiatives such as ILM, CC, AVSS, Organic standards, and ICLEI, nor has sufficient research on this been carried out for other initiatives. While for the standards, information was available on what criteria farmers have to adhere to, however, reporting on coverage of these activities was not always possible, or available. For such certification schemes, the assumption was made that these criteria are all equally adhered to, and thereby equally present in the area covered by the ICIB. In that case, a level 2 division for certification schemes focused on type of crops, as research on organically grown crops shows that the biodiversity impact is different per crop. In this case the focus on the area of crop cultivation doesn't quite capture the essence of what this study aims to explore through level 2 (i.e. changes in management practices). For the above-mentioned ICIBs, the absence of a full level 2 analysis meant that a level 3 analysis — connecting MSA values to managed areas — remained difficult or even impossible, at least for the time being.

For the Rewilding Europe case study, a ratio of area-based management categories data was directly provided by Rewilding Europe. Therefore, level 2 was available. In this case, however, level 2 also provided an interesting and specific challenge related to the practice of increasing connectivity between two areas by creating a corridor. While the area of the corridor was the area reported by Rewilding Europe, such a corridor is aimed at influencing biodiversity levels in the areas that become connected. So in this case it could be beneficial to look at both the area under the initiative, as well as the area directly influenced by such management interventions. Such an example illustrates that for some management practices, an area-based MSA assessment is quite straightforward, while for other practices it could be beneficial to look at wider impacts outside of the implementation area.

Without level 1 and 2 data being available, level 3 cannot be calculated for the case studies. For Rewilding Europe, only a part of level 3 could be calculated due to methodological limitations addressed in the next section. For two case studies, we were able to estimate its impact on biodiversity up to level 4 of the assessment (CFM and forest certification). For the other case studies, similar estimates can be made only when more data becomes available. However, even if the issue of data availability is solved, there are still methodological challenges that should be taken into account.

11.3 Methodological challenges

The methodology used in this report provides a useful framework to estimate the on-the-ground biodiversity impacts of ICIBs in a coherent, consistent way, provided that all data for the four levels are available. However, several methodological challenges exist. Some are case-specific and are described in the corresponding chapters. However, two generic challenges emerge for this methodology: (1) different types of initiatives, (2) some limitations inherent in the MSA indicator.

The eight case studies presented in this report range from landscape-wide, to management-specific initiatives. They have different approaches and baselines. Whereas some initiatives focus on specific commodity or ecosystem related practices, like agricultural products or forest management, others focus on a more landscape-scale integrated approach of managing or restoring several types of ecosystems, such as ILM and Rewilding Europe. This means that the MSA calculations either include one specific activity or a combination of multiple activities. For instance, while FSC assessments are relevant purely to forest management, ILM can for example cover changes in management practices of a forest, a grassland and a city in a specific landscape. Such a difference does not pose a problem to applying the assessment framework as such, nevertheless, it is important to keep in mind that such differences exist when analysing a certain ICIB.

Furthermore, the coverage of actual initiatives within a given area of an ICIB can be very different. Rewilding Europe, ILM, and ICLEI, for example, can cover large areas, but only induce management changes in certain parts. Subsequently, a big difference in outcome between level 1 and level 2 can be observed, as the Rewilding Europe case study shows. This does not mean that the remaining area is not influenced, since indirect impacts on biodiversity can take place there as well, such as migration of animals and the growth of non-native species of different flora. However, such indirect impacts and effects of levels 1 and 2 are beyond the focus and scope of this report. CFM, CC, and AVSS show a different pattern. In those cases, the total land area which is reached is also the area in which changes are induced directly. So for example, it is assumed that when a community managed forest increases biodiversity, this increase will apply to the entire area. Therefore level 1 and level 2 outcomes are comparable in those cases. These differences between certain cases do not rule out the possibility of an assessment for either of the cases, but does show the importance of data availability in level 1 and 2 in the analysis, namely analysing which part of the reach can potentially be biodiversity positive, and what the ratio and coverage are of distinct management practices.

A second issue, affecting level 3 analysis, is the absence of MSA categories in GLOBIO 3.6 for certain specific management practices. While for some of those MSA figures are available, such as for reduced impact logging, for others this information is not (yet) available. Furthermore, for all cases, it seems impossible to make the level 2 division in terms of different management practices and their corresponding area coverages. Many such activities potentially have a positive impact on biodiversity, but suffer from a lack of reporting of categorical data on their coverage. Moreover, this wish for more data reporting becomes particularly more nuanced and challenging as the impacts of biodiversity could quite often be indirect, but important nonetheless. For example, the criteria of Organic certification as shown in **Table 3.1** cannot always be expressed in hectares. The assumption could be made that these activities equally take place on different types of certified lands. In that case, it might be more useful to split the total area found in level 1 into either climatic zones or crop types. In the organic case, for example, a meta-analysis was presented that showed differences per crop type in species richness after organic certification. So if the total area for Organic certification is known, differences in MSA value increase could be based on the different crop types and their area coverage.

An additional methodological challenge was how initiatives were selected for the case studies. Our search methods were biased towards institutionalised initiatives that are more easily found on the internet and might have more data readily available. Selecting initiatives that are more accessible in this way inherently means that less or non-institutionalised initiatives are more easily ruled out from assessment. This study does thus encounter limitations in the case selection of ICIB initiatives due to its high dependence on already existing information and data in the public domain. Generating new data or identifying ICIB initiatives beyond what is available in the public domain is outside of the scope of this study. However, it is important to note this specific limitation of this study as far as non-state action for biodiversity conservation is concerned.

In almost all cases, examples of avoided loss can be found; for example, in the cases of certification schemes that include ‘high value conservation’ areas, rewilding projects that protect certain lands by agreements with landowners, and communities that possess territories that are sacred natural sites. In these cases, there was no change in management practice that caused an increase in MSA values, but rather it *prevented a decrease* in MSA by the ongoing protection of these lands. Thus, this might not speak directly to an increase in level 3 biodiversity impact, although the assessment framework is still able to assess the 'avoided loss principle', demonstrating that it does affect biodiversity, as a few case studies show.

Lastly, it is good to note that our assessment framework is not inherently and necessarily connected to the MSA indicator. One could think of other biodiversity indicators to use in the methodology, such as species richness, habitat quality, connectivity, etc., provided that data or models are available to estimate these indicators in relation to the interventions. Using multiple, complementary biodiversity indicators would allow for a more comprehensive evaluation of the biodiversity gains associated with the initiatives and for accommodating the plurality of values of nature.

11.4 Other limitations

This study is limited in its scope insofar as it does not assess the impacts and effects of ICIB initiatives beyond biodiversity conservation. For the Organic Agriculture case, for example, there can be a significant yield gap on the implementation and adherence of organic standards (Seufert, 2019; IFOAM, 2017; Arbenz, Gould, and Stopes, 2016). This means that more agricultural land is needed for the same result. This can often be overlooked through the application of a biodiversity specific indicator such as MSA wherein crop yields are not taken into account. In other words, changes in management practices can have positive impacts on biodiversity, but they can come with trade-offs, such as lower income for farmers or lower production numbers. It is worth noting that such trade-offs may have direct consequences for biodiversity impacts in the long run. For instance, losses of crop yield could compel farmers to revert back to non-organic practices to a greater extent or to enlarge their agricultural area into biodiversity-rich habitats. This might especially be the case when farmers are not compensated or supplemented for their respective crop loss.

Another (positive) example of indirect impacts is effects that might take place in the surrounding areas of the initiative. When an area is effectively rewilded, for example, this has an impact on the natural system outside of this area as well (Oliveira-Santos and Fernandez, 2010; Martin, 2005; Hall, 2014; Corlett, 2016). Plants might be able to recolonise surrounding areas and animal populations might become healthier (Navarro and Pereira, 2015; Corlett, 2016). Since MSA only focuses on the area where the management changes are implemented, these wider effects are not taken into account. It is worth noting that this is not an inherent limitation of MSA per se. However, it is the limitation of its implementation instead. However, as we have seen that systematically assessing the impact in the areas where practices were implemented is already quite a challenge, assessing any indirect impacts on a large scale will most likely not take place in the near future.

Lastly, another indirect impact of ICIBs is the influence they can have on governments, civil society and businesses to strengthen their efforts to protect and restore biodiversity. This impact is difficult to trace and to assess quantitatively. For example, in order to realise new nature areas, formalisation by governmental authorities is a crucial precondition for effectiveness, since they might own the land, or design and monitor the regulations for land use. In other words, collaborating with governments one way or another is often required for ICIBs, at some stage especially when upscaling. Often, governments become part of ICIBs over time through supporting, facilitating and regulating roles, despite the initiative being initiated as a fully non-state action.

For next steps in researching the impact of ICIBs, the following recommendations seem valuable: firstly, it is important to look at different types of ICIBs since they imply different techniques for data gathering, and may need to be assessed (more) context-specific depending on the type of activities carried out and information available. Moreover, characteristics of each type come with specific challenges with regard to conceptualising impact as well as with regards to methods and techniques to assess impact. Secondly, it is important that more quantitative research on the biodiversity impact of ICIBs is carried out, and that the literature is also more thoroughly screened for meta-studies on impact data, since an in-depth literature research was not possible for all case studies in the context of this report. And finally, the assessment framework could be further fine-tuned in terms of criteria, indicators and procedural steps per different type of ICIB, and its corresponding characteristics as described in the chapters.

11.5 Conclusion

First of all, this study shows the relevance of ICIBs for biodiversity conservation because of (1) the large global reach of landscapes and areas through the eight ICIBs under study here (100s millions of hectares) (see Table 11.1) and (2) the indication that they provide substantial benefits for biodiversity compared to conventional land management (MSA gains ranging from 0.05 to 0.5 on a scale of 0–1). Next to other, more classical, nature protection initiatives, ICIBs therefore have the potential to play a big role in biodiversity conservation. Whereas for nearly all case studies the biodiversity impacts are obvious in qualitative terms, quantifying the actual landscape-wide and intervention-specific impacts through all the levels of our assessment framework remained challenging for most cases. Some of the case studies could only be fully assessed up to level 2 or sometimes (partly) level 3, which shows that the potential exists to assess them in full when more data becomes available. The limitations per case study are discussed in the specific and respective chapters, but mainly come down to data availability and some methodological issues. Therefore, our framework seems to provide a good basis to assess the impacts of ICIBs using MSA as an indicator, but may need some adjustment depending on the type of initiative. This report demonstrates that there is an urgent need for additional studies to create more quantitative databases which would help to assess the impacts of ICIBs on biodiversity more holistically and conclusively. Moreover, this report demonstrates the need to contextualise biodiversity measures to the respective case.

Finally, non-state commitments have recently become an important part of the post-2020 Kunming-Montreal Global Biodiversity framework, and standardised monitoring and reporting will be even more necessary for non-state actions to be included in an accountability framework and for tracking the implementation of the GBF's *whole-of-society* approach. Since the new institutional framework includes some kind of stocktaking of efforts by non-state actors to contribute to achieving the Kunming-Montreal biodiversity targets, more harmonised and comparable assessment frameworks and indicators will be important for reporting on — and do justice to — the vast amount of work being done by non-state actors to preserve and restore biodiversity. Our method offers just one way of assessing impact, with some limitations. Notably, it does not include the *indirect effects* of non-state initiatives on biodiversity, both biophysical and governance related. We therefore recommend that more research and suggestions are needed on how to assess such indirect effects, to avoid a focus on quantitative data and technocratic monitoring only, as well as to avoid overlooking those indirect effects that are also needed for transformative change.

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