The Impact of International Cooperative Initiatives on Biodiversity (ICIBs)

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The Impact of International Cooperative Initiatives on Biodiversity (ICIBs)

Research report

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Research Report

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Executive Summary

This report deals with the impact of so-called 'International Cooperative Initiatives on Biodiversity' (ICIBs); initiatives of private, non-state actors besides, or with, national governments, and their effects on biodiversity conservation and enhancement on the ground. Five cases were analysed (alphabetical order): (1) Citizens' initiatives (CIs) that contribute to private nature conservation; (2) Community forest management (CFM) that strives for the improvement of rural livelihoods, forest conditions and forest biodiversity; (3) Landscape and forest restoration (LFR) under the Bonn challenge; (4) Re-naturing cities (RNC) for green infrastructures and biodiversity enhancement in urban areas; and (5) Voluntary Sustainability Standards (VSS) which – through market certification schemes – aims at enhancing the sustainable use of natural resources. In order to assess impact, the 'funnel-shaped assessment framework' was developed that exposes three levels: (a) number of projects or number of conservation and sustainable use areas realized by ICIBs ('outcome'), (b) positive biodiversity effects attained in certain areas, or in terms of higher 'mean species abundance' (MSA) figures, or in terms of both ('overall impact'), and (c) examples of positive biodiversity effects in specific cases on the ground ('detailed impact').

This is an explorative study applying secondary analysis of data from the literature. This turned out to be a limiting factor. For most cases, only output figures (policies, targets, etc.) – or very uncertain outcome figures could be retrieved (CIs, LFR, RNC). Only for CFM and VSS, reliable outcome figures were available in the literature. Together, these two sustainable use initiatives cover a forest area of nearly 750 million Ha. globally. Compared to the current size of formal 'Forest Protected Areas' (FPAs) of about 650 million Ha. worldwide, this figure is quite impressive.

Concerning level 2, the actual impact of ICIBs, only one case offers such data, namely CFM, for which several meta-studies are available in the scholarly literature. This literature shows that about 35% of CFM initiatives (hence, about 125 million Ha.) actually produces positive biodiversity impacts. If this benchmark would also be valid for VSS (FSC/PEFC), which is still a wild guess, then about 260 million Ha. of forests managed through CFM and forest certification perform well in terms of biodiversity impact. Taking FPAs as benchmark (650 million Ha.), such would produce an 'additionality' of sustainable use areas in which positive biodiversity impact is realized of about 40%.

While executing our study, we also decided to add a qualitative biodiversity indicator, besides quantitative ones (surface, hectares), at impact level 2. Building upon earlier work of the PBL Netherlands Environmental Assessment Agency, we adopted the MSA (Mean Species Abundance) indicator as a proxy for biodiversity. Taken the forest cases together (CFM, LFR, VSS), ICIBs are realizing an increase of biodiversity with about 15 to 25%, compared to conventional forestry.

Besides direct impact, ICIBs can also perform through indirect impact. They can influence governments, international organizations and/or nature conservation organizations to protect more biodiversity or to protect it more strictly. This type of impact is particularly relevant for citizens' initiatives, but potentially important for forestry community organizations, restoration initiatives, cities and certification bodies as well.

Level 3 of the assessment is about specific cases on the ground. We refer the reader to the individual chapters of this report to learn more about these examples.

In hindsight, we have been looking at two different types of ICIBs. Type-I ICIBs can be typified as top-down. To a certain extent, they resemble more traditional, government-led biodiversity conservation approaches, in which higher scale institutions agree on aims and methods, after which efforts towards implementation are trickled-down (see for example LFR and VSS). Type-II ICIBs can be typified as bottom-up. These are practices at the lower level of scale, ranging from communities (CIs, CFM) to cities (RNC). Focus of these practices is on local outcomes and impacts.

To conclude, this study first shows that ICIBs are very relevant for biodiversity conservation and its sustainable use, besides (inter)governmental initiatives and those of 'classical' nature conservation organizations. The report estimates – under certain assumptions and with substantial uncertainty – that ICIBs' additionality in forest biodiversity impact amounts to about 40% in quantity (size of sustainable use areas with positive biodiversity impact, compared to formally protected forest areas) and about 20% in quality (increase in biodiversity/MSA compared to conventional forest management). Secondly, this study has used and adjusted an interesting framework – 'the funnel' – to assess the outcome and impact of ICIBs. However, this framework needs further streamlining in terms of criteria, indicators and procedures in follow-up research.

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1 Introduction

1.1. Biodiversity governance

Despite many public policy initiatives to halt the loss of biodiversity at national and international levels, the results of those policies are generally insufficient to meet internationally agreed goals and targets (sCBD, 2014). Although ever more areas have become legally protected – up to about 15% of the Earth's terrestrial surface and about 5% of the marine surface – biodiversity policies were not able to reduce further fragmentation of valuable nature areas, soften environmental pressures upon them and prevent further decline of threatened species. On top of that, many governments have partially withdrawn from public policy domains, following neo-liberal ideologies, and have substantially cut on public expenses, including those for nature conservation (Buijs et al., 2014). Whereas much more governmental engagement with biodiversity is needed to achieve the internationally adopted public objectives and targets, such as those of the 2050 Vision of the Strategic Plan for Biodiversity of the Convention on Biological Diversity (CBD) and the so-called 2020 Aichi targets, policy practice seems otherwise. Therefore many observers, scholars and activists shift attention (and hope) from the governmental domain towards private and civic sectors through which public goods and aims should be guaranteed and attained (Hajer, 2011).

Such a shift also matches the governance literature very well (Peters and Pierre, 2000; Pierre, 2000). In this literature, it is suggested that due to processes such as globalization, decentralization and democratization a partial relocation of power and authority from the nation state to private and civic sectors has been taking place. Indeed, looking at nature conservation policy in the Netherlands, for example, the state has partially withdrawn from this domain, implementing substantial budget cuts, while Dutch companies and citizens show stronger direct engagement with biodiversity than before (Matthijsen et al., 2016). Besides, new 'green' governance arrangements between public and private actors have also emerged in the Netherlands as a consequence (Arnouts et al., 2012). And similar developments have been identified in other Western countries (Molin and Konijnendijk-Van den Bosch, 2014; Soma et al., 2016). Question is, though, whether biodiversity will be effectively protected and sustainably used through these new private and public-private governance arrangements.

1.2 International initiatives

At the international level, we also observe the emergence of so-called 'International Cooperative Initiatives' (ICIs) (PBL, 2015) on biodiversity, besides intergovernmental treaties like the Bern Convention, CITES and the Convention on Biological Diversity (CBD). Examples are the Bonn challenge that aims at landscape and forest restoration efforts, the net zero deforestation initiatives and 'rewilding Europe'. Yet the question remains whether and to what extent these privately-initiated ICIs contribute to international public objectives and targets, like biodiversity conservation, sustainable use and equitable access and benefit sharing. And what are the size, potential and impact of such initiatives? If we accept that this shift from government to governance is indeed taking place, to what extent do public biodiversity policies and private ones compete or complement? And what are smart (international) government strategies in a context of governance? Such questions, though, can only be answered if we are much better able to grasp the impacts of current 'ICIs on Biodiversity' (which we will abbreviate as ICIBs in the rest of this report), and whether and how they contribute to attaining international biodiversity objectives and targets.

Given the above themes, this study is very much in line with an emerging literature on ICIs on climate change mitigation, including PBL's recently published policy brief Climate Action outside the UNFCCC (PBL, 2015). This report claims that ICIs, such as the Carbon Disclosure Project (for large companies to set climate targets), the C40 initiative (for cities to set climate targets) and the Global Fuel Economy initiative (to make cars 50% more energy efficient in 2030), contribute to future greenhouse gas emission reductions to an extent comparable to current pledges of all countries around the world, as stated in their Nationally Determined Commitments (NDCs) under the Paris Agreement of the UNFCCC. Hence, the future impact of the private and civic sectors to combat global warming is impressive (although insufficient to reach the 2°C target, let alone the 1.5°C target). Other studies however include both higher (UNEP, 2016) and lower (Michaelowa and Michaelowa, 2016) estimates of the contribution of ICIs to the realisation of climate mitigation, thus indicating the methodological challenges of quantifying the contribution of ICIs and climate action outside the UNFCCC. In parallel to these studies aiming at quantifying ICI efforts, a political science literature on non-state action in the context of the climate regime also exists, exploring the implications for the further development of the climate change regime and its performance (Arts, 1998; Betsill and Corell, 2008).

1.3 Research question

Given the argumentation in the above, the central research question of this project is as follows:

> *What is the impact of International Cooperative Initiatives on Biodiversity (ICIBs) – outside the scope of the Convention of Biological Diversity (CBD) – on the conservation and sustainable use of biodiversity?*

This is an exploratory study. The aim is to develop and apply a certain methodology for assessing the impact of ICIBs rather than reaching at definite answers. The initial approach taken in the case studies is described below and will be revisited in the concluding chapter based on lessons learned in the case studies and from a comparison across the case studies. The study focuses on ICIBs that relate to the first two goals of the international biodiversity regime – conservation and sustainable use – while non target the third – the equitable access to and benefit sharing of biological resources – reflecting the expertise from the research team.

1.4 Impact assessment

In this study ICIBs are considered as a 'mean', a mechanism to achieve public biodiversity objectives and we are hence interested in the (potential) impacts of ICIBs in terms of the CBD-objectives and Aichi-targets. Assessing the performance of biodiversity action outside the CBD is much more complicated than for climate mitigation action outside the UNFCCC. Whereas the PBL climate study was assessed on the basis of one unit – greenhouse gas emission reductions, as pledged for or targeted by state and non-state actors – biodiversity impact is of a different nature due to its cumulative character (conservation and sustainable use of species, habitats, genes, ecosystems, biomes, the Earth....). In this case, we need to think of other indicators, like the number of species protected, recovered or re-introduced due to ICIBs or the size of protected areas or sustainable use areas established. Hence, one indicator, like in the case of climate, will probably not suffice. Moreover, 'real' biodiversity impact is probably very difficult to measure, given the complexity of systems, drivers, actors, effects, etc. Therefore we need to look for proxies for impact and to look at other effects in the 'implementation chain' that are more easy to assess.

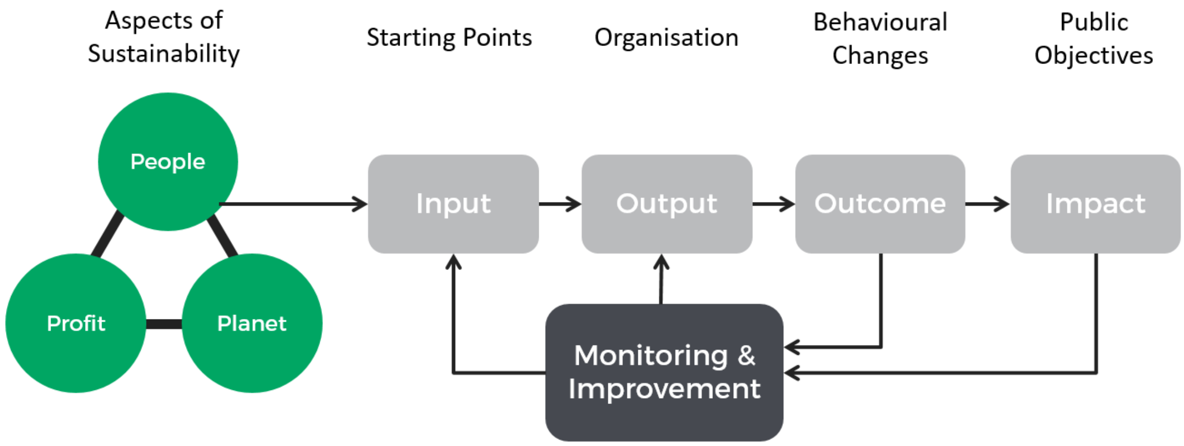


Figure 1.1 *Implementation Chain - Assessment framework for the sustainable development of supply chains (source: Van Tulder, 2010; adaptation by PBL*

Figure 1.1 presents a model of the 'implementation chain' used in another PBL study (van Oorschot et al, 2014). The figure tells us that decision-making, for example on sustainability, needs 'inputs' (ideas, agendas) and produces a number of effects, however not only 'outputs' (organizations, policies), but also 'outcomes' (behavioural change by target groups) that ultimately should result in 'impacts' (attainment of policy goals on the ground). The latter is checked through monitoring. However, the impact of ICIBs may not be related to their own outcomes only, but to ones of other actors as well. For example, their efforts may stimulate governments to increase efforts on biodiversity conservation, they may contribute to the effectivity of governmental systems to use biodiversity more sustainably or their protest against economic development projects potentially harming biodiversity may pressurize governments or industries to redesign their projects or even abandon these (Herrfahrdt-Pähle and Pahl-Wostl 2012). Many such indirect effects relate to ICIB's engagements in polycentric governance systems in which non-state actors complement governments (Biggs et al. 2012; Buijs et al. 2016). These indirect effects are visualized in *figure 1.2* below.

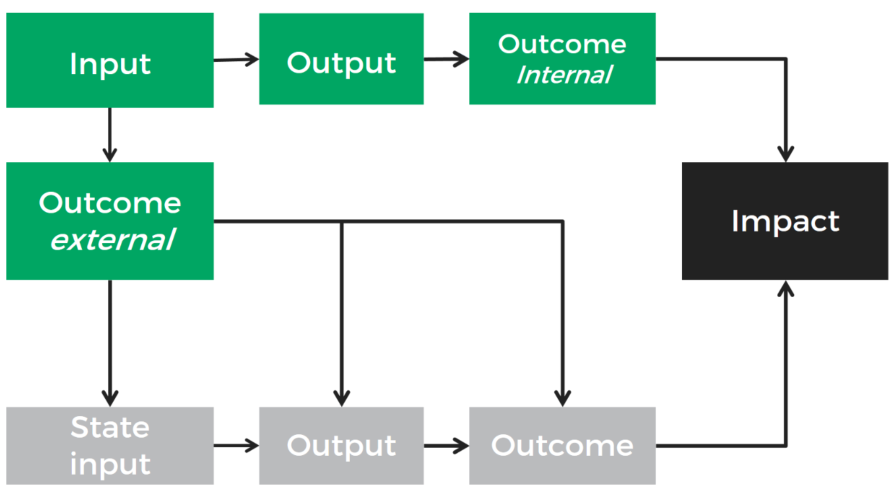
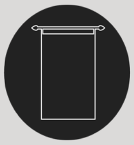


Figure 1.2 *Indirect Effects on state action*

Our assumption is that each ICIB has at least the ambition to produce (indirect) effects throughout the entire (or at least part) of the implementation chain, and that it also has certain ideas of how to perform these, while positioning itself in the implementation chain. The latter we call 'the theory of change', so how initiatives assume that they will attain their goals, not only on paper, but in practice as well. In this report, we will analyse the impact of a number of ICIBs in the next chapters, using this implementation chain scheme as a basis, while reconstructing their theory of change. *Figure 1.3* shows the set-up plus the symbols for how we will do this (with the example of forest landscape restoration).

Aim of ICIB for example to restore landscapes in order to restore ecological integrity as well as improve human well-being. Pledges, commitments (e.g., Bonn Challenge), partnerships (e.g., GPLFR), funding and agenda setting of landscape restoration (e.g., GLF)



Project and programme design, national action plan, action strategies etc.

Implementation of (direct effects) or influence in (indirect effects) programmes and projects on the ground.

Actual direct or indirect impact on diversity

Figure 1.3 *Elements of a 'Theory of Change'*

This study is mostly interested in impact, as the research question shows, but it is already acknowledged in the above that assessing impact is a complicated job (Murray et al., 2010). Besides the 'problem of attribution' (whether an effect can be causally attributed to a certain intervention; Arts & Verschuren, 1999; Ton, 2012), or the 'issue of contribution' (how one factor among many others contribute to an impact; Arts and De Koning, 2017; Ton et al., 2014), it is very well possible that data for the overall assessment of biodiversity impact will be hardly available for various ICIBs, so that we might have to rely on achievements in earlier stages of the implementation chain (like outcome, or even output). Yet, we will try to come as close as possible to the impact side of the implementation chain for all ICIBs included in this study. However, we will do so through secondary analysis. In other words, we do not assess impact ourselves, but rely on existing studies which measure effects of the ICIBs in this report. As a consequence, we also have to accept the methodological approaches of these studies and their degree of 'rigour' concerning impact assessment (Ton, 2012).

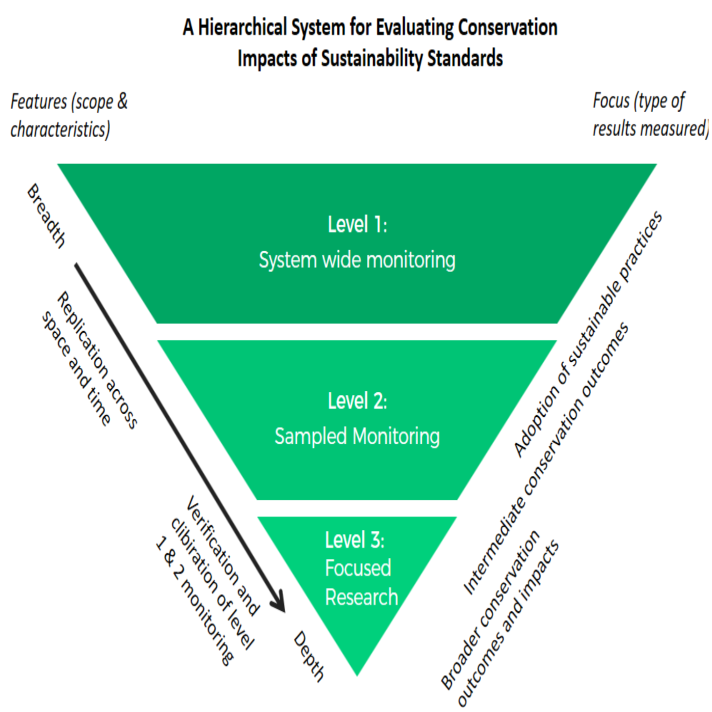


Figure 1.4 *Three-level model to evaluate conservation impacts of sustainability standards (Milder et al., 2015)*

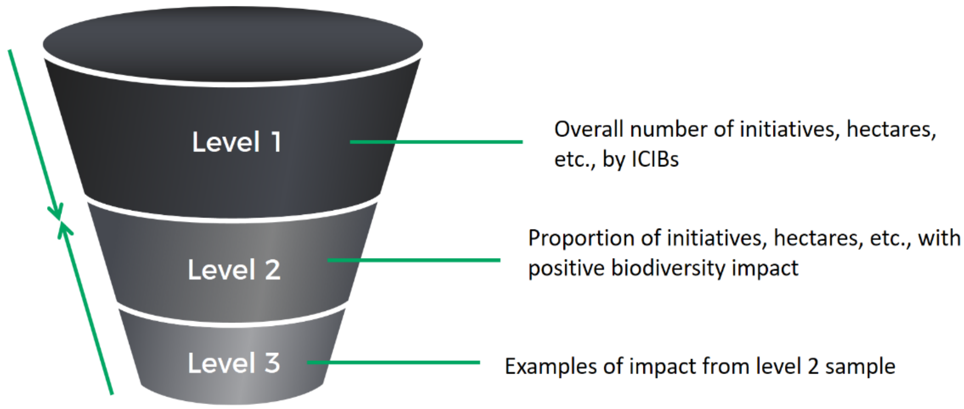


Figure 1.5 *Funnel-shaped assessment framework*

In order to structure and process data of various studies, we designed a framework, which is very much inspired by the work of Milder et al., 2015 (see *figure 1.4*). We however simplified the model to what we call the 'funnel-shaped assessment framework' or simply 'funnel' (see *figure 1.5* above). Milder et al. (2015) and colleagues propose an approach for system wide-monitoring and evaluation of the conservation benefits of Voluntary Sustainability Initiatives (VSS). It provides a three-level approach that ranges from system-wide monitoring (level 1), to sampled monitoring (level 2) to in-depth focused research (level 3) that verifies and calibrates level 1 and 2 monitoring. We apply the same logic to identify indicators for outcomes and impacts of ICIBs and infer estimations of the latter based on combinations of system-wide monitoring (level 1) and in-depth insights in impacts (level 3).

In the first top layer of the funnel, data refer to 'outcomes', for example overall number of projects implemented, number of hectares of protected areas implemented or the size of sustainable use areas realized by ICIBs. These overall data are often available. The third, bottom level of the funnel refers to identified positive biodiversity impacts of single initiatives. Such in-depth data are regularly available as well, through in-depth case studies. But level 2 information, the overall proportion of ICIB's projects and programs with positive biodiversity impact worldwide, is largely unknown.

For example, we roughly know the global area of forests under community management in the world (level 1) and we also know positive biodiversity impacts of individual community forestry projects, thanks to in-depth case studies for impact assessment (level 3) (see Chapter 3). But we do not have a general overview of positive biodiversity impacts of community forestry policies, projects and programs world-wide. And that's exactly the type of knowledge we would like to acquire for this study: the total sum of hectares of community forestry with positive biodiversity impacts on Earth. With this aim in mind, we enter into level 2 of the funnel. Now, the idea is that we deduce knowledge on level 2 either through 'specification' (level 1 --> level 2) or through 'upscaling' (level 3 --> level 2). 'Upscaling' can be done by making an educated guess about whether the biodiversity-positive sample of level 3 can be generalized to level 2, or whether some kind of correction factor needs to be included, and 'specification' by making an educated guess about the proportionality of biodiversity-positive cases as compared to all of them.

In addition to area (Ha.) with positive biodiversity impacts, some of the chapters that follow will also try to assess the degree of this impact caused by ICIB interventions. In so doing, the indicator of MSA ('mean species abundance') that is used in GLOBIO assessment studies, including those of PBL, will be applied

(Alkemade et al, 2009; Schippers et al, 2016; www.globio.info). MSA indicates the average ‘naturalness’ of more or less managed systems (e.g. intensively logged forests versus selectively logged forests versus natural forests). By inducing a transition from one system state to another, due to an ICIB intervention, average MSA gains – or prevented MSA losses – can be calculated. Details of this approach will be elucidated in the respective chapters.

Finally, some of the chapters that follow will also assess the additionality of ICIB initiatives compared to what governments are currently doing in the context of the CBD (and other biodiversity treaties). In order to do so, we particularly take the monitoring reports related to the CBD as baselines: the fourth Global Biodiversity Outlook (GBO-4), its background reports (sCBD, 2014; Leadley et al., 2014) and the review of NBSAPs and national reporting (Pisupati & Prip, 2015).

1.5 Case Studies

Since this is an explorative and secondary study, we chose ICIB cases that are diverse in nature (more or less linked to governments, having direct and indirect impacts, recently established and older, bigger and smaller in size) and of which (at least some) data are available in the literature. And since this is also a study with a rather limited budget, we chose case studies that are close to our recent work (both PBL and FNP studies). The following were selected: (1) landscape and forest restoration (LFR) efforts, as endorsed by the Bonn challenge; (2) Voluntary Sustainability Standards (VSS), particularly forest certification, which aims at enhancing the sustainable use of forest resources; (3) Citizens’ initiatives (CIs) in Europe that contribute to nature conservation; (4) Community forest management (CFM) that strive for the improvement of rural livelihoods and forest conditions, particularly in the Tropics; and (5) re-naturing cities (RNC) for green infrastructures and the enhancement of biodiversity in urban areas.

The remaining of this report consists of case study chapters (in alphabetical order; CIs, CFM, LFR, RNC, VSS) and a final, concluding chapter.

2 'Citizen Initiatives' (CIs)

2.1. Introduction to Green Citizen Initiatives

Citizens and local communities have a long history of engagements with and contributions to the conservation of biodiversity and natural areas (Koppen and Markham 2007). Recent shifts towards multi-level, collaborative, and polycentric governance combined with local has once again focused the attention on contributions from citizens and societies to biodiversity conservation (Buijs, Mattijssen, et al. 2016). The governance of urban and non-urban green has shifted towards a multi-level, collaborative, and polycentric governance approach, involving a diversity of actors and governance levels into decision-making (Fors et al. 2015; Ferranti et al. 2014). Several scholars have suggested that especially European and US cities now experience the next step in polycentric governance, where the involvement of citizens in greenspace governance further shifts from a focus on public participation in government policies towards increased active citizenship (Buijs and al. 2017; Dennis and James 2016; Andersson et al. 2014; McMillen et al. 2016; Chan, DuBois, and Tidball 2015). Indeed, Aichi Target 1 states that by 2020, at the latest, people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably. While most Aichi targets still seem out of reach, support for social change movements and for the co-governance of natural resources have been identified as possible options to halt biodiversity loss (Hill et al. 2015). For example Local Agenda 21 has stimulated local initiatives to halt biodiversity loss or contribute to local enhancements of biodiversity (Van Herzele, Collins, and Tyrväinen 2005). In an European overview, examples of citizens involvement was found in all 16 participating countries (Van der Jagt et al. 2016; Buijs, Elands, et al. 2016), although the level and extent of these examples vary significantly. While for example South-European cities showed predominantly many examples of citizen involvement in urban agriculture, the diversity in the UK, Germany, and the Netherlands was much larger, including many examples of self-governance focusing explicitly on biodiversity protection (Buijs, Elands, et al. 2016; Mattijssen et al. 2016; Lawrence and Ambrose-Oji 2015).

This involvement of citizens in protection, maintenance and enhancement of biodiversity is not primarily driven by government policies, but by intrinsic motivation of citizens (van Dam 2016). The majority of people across Europe acknowledges the need for biodiversity protection and support policy actions in this respect (Eurobarometer 2013). Connectedness to nature and the experience of nature are relevant life experience for citizens around the globe (Chan et al. 2016). People feel related to nature, they feel morally inclined to protect it, and they feel happy and healthy when enjoying nature (Tzoulas et al. 2007). Based on these moral and experiential dispositions towards nature conservation, many people are motivated to contribute to the conservation of nature and biodiversity.

Several authors argue that as a result of changing governance discourses and practices, citizen involvement has increased over the last years or decades (e.g. Lawrence and Ambrose-Oji 2015; McCarthy and Prudham 2004; Blanco, Griggs, and Sullivan 2014). The institutional and political trend towards devolution and localism (Apostolopoulou et al. 2014) is accompanied by an increase in active citizenship (Hoskins 2009) at the local level. The interest from local citizens to actively engage in the conservation of green spaces and biodiversity can be linked to an increased understanding of the benefits of greenspaces in delivering public goods (Lawrence et al. 2009); a raise of neo-communitarianism to address social inequality (Fyfe 2005); and more general trends towards co-production of public goods and services (Pestoff, Brandsen, and Verschuere 2012) in the 'energetic society' (Hajer et al. 2015).

Within this chapter, we focus on an important manifestation of active citizenship: green citizen initiatives. We understand a green Citizen Initiatives (CIs) as [a group of citizens] who organize themselves in a multiform manner, to mobilize resources and to act in the public […] in order to protect rights and take care of common goods (cf. Moro 2012, , p.11). CIs are not to be confused with citizen participation, in which citizens are invited to participate in policy making or implementation. CIs also differs from state initiated co-governance models and other forms of governance in which state actors actively seek collaboration with non-state actors. In general, CIs are not based on government aims or interventions, but inspired by the motivations of people and communities to improve their environment (van Dam 2016). These motivations often also include an inclination to contribute to biodiversity conservation. Although CIs may collaborate with governments, they usually operate somewhat independently from states and other governmental actors, in a much less institutionalised and coordinated manner than e.g. co-governance (Uitermark 2015; Crowe, Foley, and Collier 2016). Alternative concepts related to the contributions of organised active citizens to the common good refer to self-organisation (Uitermark 2015), self-governance (Sørensen and Triantafillou 2009), Do-It-Yourself democracy (Crossan et al. 2016) or bottom-up governance (Waterton et al. 2015).

Green CIs relate to a diverse set of green areas, from forests and woodlands, wild and natural places, parks, community gardens, and urban agriculture. The relationship with state actors also differs significantly between groups: from emerging forms of political resistance like guerrilla gardening or protesting (Adams, Scott, and Hardman 2013; Colding and Barthel 2013); to collective action in private and public spaces financial supported and stimulated by municipalities and NGO's(Barthel, Parker, and Ernstson 2015; Colding and Barthel 2013). Meanwhile, the emergence and success of CIs critically depends on the available cultural capital (van Dam, 2016).

CIs are located in the complex nexus of governance arrangement, socio-economic and cultural structure of countries and communities as well as pressing socio-environmental challenges. Consequently, practices of CIs differ significantly across Europe and beyond (Buijs, Elands, et al. 2016). For example, the recent financial crisis has boosted urban agriculture initiatives all over Europe. In Southern Europe and the new member states, this form of green CIs seems dominant, while other forms are less visible (Buizer et al. 2015). Meanwhile in several North-West European countries, most notably the UK, the Netherlands and Germany, the pallet of CIs is much wider, including numerous CIss that strongly focus on biodiversity protection and/or monitoring (Lawrence and Ambrose-Oji 2015; Buijs, Mattijssen, and Elands 2016).

2.2. Motives, goals, targets

Motives

Contrary to State policies, the motives and goals of CIs are not based on overarching national or international aims and agreements, but on the individual motivations of participants. Their engagement with biodiversity and green spaces can be motivated by a combination of social and environmental objectives, rooted in a type of environmental stewardship that goes beyond immediate personal benefit (Krasny et al. 2014). The motives to initiate or engage in green initiatives are usually based on motivations based on intrinsic values, self-enhancement, and social interaction (De Groot, Salverda, Donders, et al. 2012).

Goals

CIs generally pursue several goals:

- Enhance and protect natural areas in protected areas, cultural landscapes and urban green areas
- Improve the use and experience value of these areas
- Raise awareness and knowledge on nature conservation
- Contribute to social cohesion and empowerment

Based on Dutch examples (Mattijssen, Buijs, et al. 2017), approximately two-third of all initiatives explicitly aim to enhance and protect natural areas. Although sometimes this is explicitly focused on biodiversity protection, usually it is focused on the protection of green areas in general. In addition to these environmental goals, also social and community goals are formulated, related to social cohesion, education and recreation.

Targets

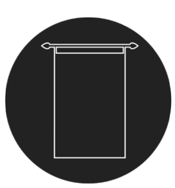
CIs are usually much less formalised and institutionalised compared to other non-state actors and institutions (van Dam, Salverda, and During 2014). Consequently, they tend not to formulate specific targets on which they can (or want to be) held accountable for.

2.3. Theory of change: from input to impact

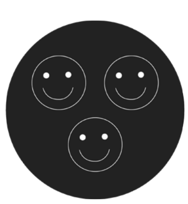
To assess the possible contributions of CIs to halting biodiversity loss, we need to slightly alter the input-output-outcome-impact model. To fully appreciate the outcome and impact of CIs, we need to distinguish between direct and indirect outcomes and their contributions to actual impacts on biodiversity (see *figure 1.2*). Direct outcome of CIs relates to the direct outcomes of specific CIs. For example CI Penllergare Trust in the UK contributes to the conservation and restoration of 100 ha of natural area and its biodiversity and recreational values in Wales (Pennlergare 2016). This output is a direct result of their efforts to find the financial and human resources to manage this area.

In addition to these direct outcomes, CIs may also contribute to the effectivity of 'external' governance systems to protect biodiversity (Herrfahrdt-Pähle and Pahl-Wostl 2012). We call this the indirect outcome of CIs (see f*igure 1.2*, Chapter 1, p.3). CIs focusing on political actions may stimulate governmental actors to increase efforts regarding biodiversity protection or avoid or revise developments potentially harming biodiversity. Several examples exist in which CIs managed to halt specific threats to natural areas from planned urban developments or infrastructure projects. Usually this impact on biodiversity is thus indirect by influencing governmental policy. These indirect outcomes relate to engagements in polycentric governance system where active citizens complement other actors by providing local knowledge or access to specific actors, amongst others (Biggs et al. 2012). Citizen groups often have better access to local communities, and can thus help making biodiversity governance more inclusive (Buijs, Elands, et al. 2016). Finally, environmental education and increased connectedness to nature among citizens may in the long run contribute to enhancing public support for biodiversity protection schemes from state-actors.

Input: The input of CIs exists of efforts by motivated citizens who aim to contribute to environmental and/or social improvements. Social and cultural capital of participants and the wider local community are usually critical factors for success of CIs. Financial resources differ significantly among CIs. Although many CIs manage to find financial resources outside governments, subsidies and other financial support from governmental bodies remain important. Access to and decision making power over public land are additional important resources, and therefore governments often remain an important actor (Colding., et al. 2013).



Output: Formal output, such as formalised plans, are rare for CIs. Only large and more institutionalised CIs may produce such formal plans to structure their activities or to find external financial recourses. However, in general, producing output is not the primary aim of most CIs, and depending on their formal status, many CIs do not need to acknowledge their output for external partners.



Outcome: CIs contribute to a diversity of environmental and social outcomes. In addition, these outcomes can be direct or indirect (see *figure 1.2*, Chapter 1, p.3). Direct environmental outcomes relate to the production of additional greenspace, or the enhancement and maintenance of existing greenspace (Colding et al. 2013; Mattijssen et al. Draft). Reported benefits of active citizenship include the provision of regulatory ecosystem services (Krasny et al. 2014), an increase in biodiversity and pollination (Dennis and James 2016) and a decrease in CO2 emissions (Barthel, Parker, and Ernstson 2015).

Next to environmental outcomes, a diverse range of direct social outcomes, or co-benefits of CIs may also occur, related to for example the social cohesion of communities and societies, including the ability of groups or communities to cope with external stresses and disturbances as a result of social, political, and environmental change (Adger, 2000, p.347). By bringing people together and formulating shared goals, citizen groups are known to contribute to social cohesion (Veen 2015). Citizens often also benefit from improved self-organisation as a result of capacity building in areas such as food growing (Bendt, Barthel, and Colding 2013), and the promotion of environmental awareness and education.

CIs may also have indirect outcomes. These are the outcomes through which CIs as non-state actors may contribute to the input, outcome or impact of state-actors. Most notably, CIs that engage in political actions to stop specific biodiversity threats through for example planned urban or commercial development in natural areas can realize impacts upon the behaviour of state. CIs may also contribute to the successful implementation of biodiversity policy.

Impact: The impact of CIs on biodiversity will differ significantly between initiatives. Some CIs explicitly aim for biodiversity impacts, while for others focus will be more on social impacts, such as recreational use of green areas. These CIs may have an ecological impact, but this often is not the case. Sometimes impact may even be negative, when green areas are put under more extensive management schemes to increase the use value of the area.

Box 2.1: CI's theory of change

Input: Motivated active citizens, social and cultural capital

Output: CIs usually do not produce formal outputs

Outcome: CIs may have direct and indirect outcomes. Direct outcomes are the production of additional greenspace, or the maintenance of existing greenspace. Indirect outcomes contribute to the input, outcome or impact of state-actors.

Impact: Depends on type of CIs and ranges from benefits such as biodiversity conservation to co-benefits such as increased leisure opportunities, environmental awareness, and increased social cohesion

2.4. Past performance: assessing outcome and impact

As CIs are much less formalised and institutionalised compared to most other non-state actors (van Dam, Salverda, and During 2014). They usually focus more on actual activities than on formulating explicit goals, tasks and strategies. Moreover, no overarching national, European or global organisation or network exists that collects or analysis such data. There are specific topics on which European or global networks have evolved, such as for urban agriculture or Transition Towns, but these are loosely organised bottom-up networks without explicit efforts to quantify or monitor results. Because of this lack of institutionalisation on the national or supra-national level, it is difficult to sketch out a complete picture of the role of CIs in biodiversity protection. The emergence and effects of CIs therefore heavily depend on the national and local context, such as governance structure and capacity, physical characteristics, social and cultural capital of citizens and communities, and ownership of natural areas.

Active citizenship in general differs significantly between European countries. Based on the four dimensions of active citizenship by Hoskins and Mascherini (2009), protest and social change (including engagement in environmental organisations), community life (including engagement in social organisations), democratic values and representative democracy, we could consider participation in environmental citizen initiatives at the crossroads of protest and social change and especially community life. Analyses of the European Survey by Hoskins and Mascherini (2009) suggests that Sweden and Norway clearly stand out on ranks 1 and 2, with other North-Western countries, also score high on engagement in both protest and social change and community life, with a high score on community life for Belgium (rank 3) UK (rank 4), and The Netherlands (rank 5). Active citizenship in Southern European countries and especially the New Member States is considerably lower (Ibid).

This overall analysis of the distribution across Europe is in line with recent European case-studies (Buizer et al. 2015; Buijs, Elands, et al. 2016; Van der Jagt et al. 2016) and expert judgement, which suggests that the UK, the Netherlands and probably Germany (especially in the cities) are at the forefront of CIs involvement in biodiversity protection (Buizer et al. 2015; Buijs, Elands, et al. 2016; Van der Jagt et al. 2016).

The UK clearly stands out in a well-developed national strategy to stimulate citizens and communities to initiate or contribute to biodiversity protection, especially through forests and woodland (Reed and McIlveen 2006). Related to the recent political focus on citizens and communities as well as to longer lasting traditions of community involvement, England, Wales and Scotland have explicitly developed community forestry planning, including evaluation of these projects. Recent studies suggest there are now at least 650 community woodland groups in the UK (Lawrence and Ambrose-Oji 2015). Although most community woodlands are collaborations between non-state and state actors, non-state actors have initiated the majority of these groups[1] . For Scotland, the country with probably the highest percentage of community forest in the UK, more than 8% (55 000 of the 667 000 ha) of publicly owned forest is covered by informal or formal management agreements with local communities. A recent count of community woodland groups in Scotland found a 67% increase in the last five years, bringing the number of community groups to 204 (Lawrence and Ambrose-Oji 2015). Furthermore, National tree planting statistics show that approximately 1000 ha out of the annually planted 5000 ha during the 1990s and early 2000s were planted in Community Forestry projects (Read et al, 2009). It is suggested that this number has declined since, as woodland plantation has been replaced as primary goal of Community Forestry projects by more socially oriented aims such as empowerment and the building of social capital (Lawrence and Ambrose-Oji 2015). However, state regulations strongly focus on community groups to foster sustainability, including woodlands and biodiversity. Statutory requirements exists for every local authority to prepare a “Sustainable Community Strategy” (Local Government Act 2000). In addition to plantation, community woodland groups also significantly contribute to bringing woodland into sustainable management (Owen et al. 2008).

Level 1

The availability of quantitative data on CIs is rather limited. The most extensive overview of the quantitative outcomes of CIs is a recent study of 264 green initiatives in the Netherlands by Mattijssen et al. (2017). This study identified a large range of green CIs in the Netherlands. We focus on this study in our analyses of the outcome of CIs. Because of high uncertainties in the calculations, all numbers in this chapter need to be considered tentatively, as a very first estimate of possible outcome and impact of green CIs. Moreover, we will present the results as a range of possible numbers, dependent on the assumptions that constitute the calculations below.

The study by Mattijssen et al. clusters the range of citizens initiatives in three distinctive types: i) initiatives focusing on *restoration and management*, ii) initiatives focusing on *use* (including experience and education) and iii) initiatives focusing on *political influence.* Aims as well as impacts differs significantly between these aims, including co-benefits such as social cohesion or increased public support.

CIs in the restoration and management cluster generally aim explicitly at restoring or managing valuable natural landscapes in rural, peri-urban and urban areas. Biodiversity protection and enhancement is often

1 personal communication by B. Ambrose-Oji

included in the aims of such initiatives. In the use cluster, the main aim of initiatives tends to focus on co-benefits and increasing the use value of natural areas, including aesthetical quality, education about the natural environment and strengthening the connectedness to nature through experiencing nature through e.g. green playground or nature trails. Biodiversity protection is usually not an aim. Many of these initiatives are located in urban and peri-urban areas. The political influence initiatives also tend to focus on protection of natural areas, including biodiversity. In general, the aim is to avoid (further) loss of biodiversity or to promote government action for increasing biodiversity values. While the conservation strategy of the use and restoration and management groups focus on hands-on physical conservation activities, such as planting, pruning, or maintaining plants and paths, political influence group use a strategy of influencing institutional actors such as states, municipalities and NGO's, often to prevent (further) biodiversity loss, but sometimes also to increase investments or maintenance efforts in valuable natural sites. Such strategies may include protesting against urban developments, lobbying for protection measurements, or developing and promoting alternative plans for an area.

Although co-benefits may have an indirect impact on biodiversity (see Chapter 1 and (Bain et al. 2016), this chapter does not focus on such co-benefits, but on the direct benefits from Cis though for example contributing to the conservation or management of a natural area. As use groups generally aim at co-benefits and not at benefits such as biodiversity protection, these groups are excluded from the analysis.

The biggest challenge to estimate the total number of green CIs in the Netherlands lies in identifying all individual initiatives. Mattijssen et al. (2017) used several techniques to identify a large range of examples. Nevertheless, they acknowledged that they haven't been able to capture all existing examples in the Netherlands. In general it is likely that the inventory will be somewhat biased towards well-documented and visible initiatives, while smaller and less visible initiatives stay below the radar as they are more difficult to find (see also Uitermark, 2015).

To estimate the total number, we extrapolate the sample by Mattijssen et al. to the entire population of green CIs in the Netherlands. The only additional data set available for comparison is an independently collected sample of green CIs by Kruijt et al (unpublished). Unfortunately, this second sample doesn't use a well-defined typology and doesn't include political influence groups. Tentative comparison of the two samples suggests that the restoration and management type shows quite some overlap and that the Kruijt sample doesn't contain much additional initiatives here.

To estimate the total number of initiatives in the full population based on the Mattijssen sample, we use a multiplier for each cluster of CIs. This multiplier is based on the proportion of all CIs that is included in the sample. If for example we estimate that 50% of CIs in the Netherlands are included in the Mattijsen sample, we use a multiplier of 2 (= 1 / 0.5) to calculate the total number of CIs based on the number in the sample.

To estimate the multiplier needed to extrapolate from the sample to the full population of CIs, we use expert judgements from three researchers involved in the study of Mattijssen et al. To acknowledge for the uncertainties in this extrapolating exercise, each researcher independently estimated the maximum range for the multiplier for each type of CIs, after which these were combined. For example, for the restoration and management group, researcher 1 suggested a multiplier of 2, researcher 2 a multiplier of 4 and researcher 3 a multiplier of 2 (see *table 2.1*).

In addition to estimating the number of initiatives, also their size needs to be estimated. The sample of Mattijssen et al. unfortunately includes only a limited number of examples for which data on the actual size of the natural area is available (87 out if 264 initiatives). This contributes to the uncertainty of the calculations. While most initiatives work on small patches of 1 ha or less, the biggest example in the database tries to protect 6000 ha. Furthermore, as Uitermark (2015) has suggested, smaller initiatives will usually be less visible and documented. As such, our sample of initiatives will have a bias towards larger initiatives.

One option to account for this skewness, is combining the mean and the median[2] to estimate the size of the initiatives. As a consequence, the average size of the initiatives is displayed as a range from the median of the sample to the mean of the sample (*table 2.1*).

Total size of CIs (ha) = ∑types([range in size] * [#initiatives in database] * [range in estimated multipliers])

Level 2

However, not all CIs contribute to biodiversity conservation. Unfortunately, very little studies have investigated the impact beyond the individual case level (Buijs, Mattijssen, and Elands 2016). A cross-country review study into citizen participation in general also concluded that the majority of evaluation studies focus on process evaluation or social impact rather than ecological impacts (Fors et al. 2015). Even in the UK, with a much longer tradition of community forestry, an overview of ecological impacts seems lacking. Although a review study identified 70 evaluation studies concerning community forestry, covering 681 evaluation cases, estimates of ecological impacts are lacking (Lawrence and Ambrose-Oji 2015). Most community groups do not conduct any kind of monitoring or evaluation (Lawrence and Moltena 2012). Finally, even in individual case evaluations, the focus seem to shift from hectares to social outcomes (Ibid). In a study of Community Woodlands in Wales, (Owen et al. 2008) conclude that such woodland groups significantly contribute to bringing woodland into sustainable management.

The first group of CIs, those focusing on ecological restoration and management all explicitly aim to contribute to nature conservation. As this usually is their prime objective, we can safely suggest these initiatives will have a positive impact on biodiversity. This size of this impact ranges from installing nesting boxes up to actual ecological restoration projects of several hectares. As these groups work directly on the ground, no implementation loss will occur, as may happen with governmental policies. Consequently, we will assume a 100% impact rate for this group of CIs. This results in an estimated positive biodiversity impact for 630 - 5,460 ha.

The second group, CI's focusing on political influence, also tend to focus on a contribution to nature conservation, often through trying to avoid biodiversity loss or to motivate authorities for increasing existing biodiversity values. However, political groups do not work hands on, but focus on influencing policy and management by governments or corporations. Their impact on biodiversity protection thus depends on the success rate of these groups. Only if groups are successful in influencing decision making, these groups will have impact. Unfortunate, the success rate of political groups in actually protecting or developing green areas is as of yet unknown. If only 10% of these CIs would be successful, and thus have impact, their impact would be comparable to the impact by the restoration and management groups. As no reliable estimate exist on the rate of their, we cannot estimate the impact. This is unfortunate, as we expect their impact to be significant.

The additionality of these efforts by CI's can be compared with the 616,000 ha of designated nature conservation areas (The NNN) in the Netherlands (IPO, 2017). CIs produce outcomes on 1848 – 88,110 ha (*Table 2.1.*). That is an addition of 0.3% to 14% of the total designated area in the Netherland. It should be noted however that only 10% of CIs focus on protected areas. 90% focus at least partly on cultural or urban landscapes.

Estimated impact is significantly lower, as we do not include possible impacts from political influence groups in this calculation. The estimated biodiversity impact on at least 630 – 5,460ha by green CIs (*Table 2.1.)* then amounts to an additional contribution to designated nature conservation areas of at least 0,1% to 1%. Because of the exclusion of two out of three types of CIs, we consider this to be the minimum impact of CIs. In addition, it needs to be mentioned that the indirect effects of CIs on environmental awareness and public

2 The median is the "middle" value of the sample, separating the 50% of examples with the lowest scores from the 50% of examples with the highest scores. It is commonly used as an alternative for the mean if the skewness of a sample is large.

Box 2: Assessing the impact of CIs: methods used

CIs are bottom-up initiatives with usually a low level of instutionalisation, only limited data are available. Calculations are based on a single country study (the Netherlands) by Mattijsen et al. 2016

Step 1: We distinguish between 3 types of active citizenships: i) restoration and management, ii) use, and iii) political influence.

Step 2: The biodiversity impact of the use type (3rd) is considered to be minimal, zero or even negative. This type is excluded from the calculation

Step 3: The number and size of CIs per type is derived from the database by Mattijssen et al.

Step 4: Because the sample used stems from an unknown population of CIs, experts (N=3) have estimated the multiplier needed to estimate the total number of CIs in the Netherlands. To account for uncertainties, the range in expert judgements for this multiplier is used in the calculation. This results in a range in the estimate of the number of CI in the Netherlands

Step 5: The average size of an CI is estimated. As we expect our sample to have a bias towards larger CIs, we use the range between the median and the mean of the size of CIs in the sample as an estimate for the actual size of CIs

Step 6: We estimate the impact of the restoration and management type CI as 100%, because these are bottom-up initiatives that start doing something in nature, and not write a report first that needs to be implemented, which nearly always goes together with implementation deficits. However, we have insufficient data to estimate the success-rate of political influence CI. We therefore exclude this group from the final calculation. Consequently, the outcome underestimate the actual impact of CIs

***Table 2.1** Outcomes and impact of green CIs in the Netherlands (ha)*

Type	Number of initiatives in dataset [a]	Multiplier [b]	Estimated total number CIs in NL [c]	Estimated average size (ha) [d]	Estimated outcomes: Total size of all CIs (ha) [e]	Estimated biodiversity impact [f]	Estimated impact (ha) [g]
Restoration and management	105	2-4	**210-420**	3.0-13	**630-5,460**	100%	**630-5,460**
Political Influence	87	2-5	**174-435**	7.0-190	**1218-82,650**	Significant but unknown	**Significant but unknown**
Total	192	-	**384-855**	-	**1848-88,110**		**At least 630 - 5,460**

[a] Total number of initiatives in dataset

[b] Multiplier to account for incomplete dataset. The range is based on expert judgement by 3 researchers , all involved in (Mattijsen et al, 2016)

[c] Estimate of total number of this type of CIs in the Netherlands, computed as [the number of initiatives (a)] x [range in multiplier (b)].

[d] Estimated average size. To account for uncertainties as well as possible skewness, this is described as a range from median to mean for this type of CIs

[e] Estimated total size of all Cis in the Netherlands with possible impact on biodiversity. Computed as [estimated total number] x [estimated size]

[f] Estimate of actual impact. For Restoration and Management groups we estimate the impact 100%, for the political influence groups actual impact is unknown

[g] Estimated total impact on biodiversity in the Netherlands. As the impact of political influence groups is unknown, this group is not included in the calculation

support for governmental policies probably outweigh the direct outcomes of green CIs in protecting and managing natural areas. After all, increased environmental awareness and public support may contribute to more ambitious policies and more effective policy implementation (Bain et al. 2016).

Extrapolation from the Dutch context to a European context is very difficult. Although the terminology self-governance suggests a relative independence from governmental organisations, in practice CIs are often embedded in existing governance structures and depend strongly on the willingness to collaborate from state, municipal or agricultural land owners (Buijs, Elands, et al. 2016). Governance structure, ownership and land use differ significantly between countries. This especially holds for forests. The Dutch context of predominantly large State forests in combination with large Environmental NGOs focusing on nature conservation instead of timber is hardly comparable with private forestry that focuses more on timber production in many other European countries. Meanwhile, community owned forests are relatively common in e.g. the UK, while very rare in the Netherlands. Furthermore, compared to most other European countries, the Netherlands is characterised by urban and peri-urban landscapes, with relatively few truly rural landscapes. This has significant impacts on the demographic reservoir of citizens living near such landscapes, that is the sheer number of people that could get involved. As such, a truly European or global overview of the contribution CIs lies far beyond the horizon. And although Aichi Target 1 states that by 2020 “People are aware of the actions they can take to conserve and sustainably use biodiversity”, data are unavailable and indicators still have to be developed (BIP 2017).

Level 3

Here we present two examples of possible outcomes and impact of CIs across Europe: De Ruige Hof, Amsterdam and Penllergare Valley Woods

De Ruige Hof, Amsterdam

Nature association De Ruige Hof (The Wild Court) manages 13 hectares of nature in the southeast of Amsterdam. Local citizens initiated the association in 1986 to protect spontaneously emerging nature on abandoned construction sites. Over the last 30 years, local governments launched several development projects for the area, including housing and a new highway. The activities of the Ruige Hof, its monitored value for biodiversity and their status as a well-known recreational area has thus far prevented the implementation of such projects.

Next to preventing urban development in the area, the aim of De Ruige Hof is focused on increasing human-nature interactions as well as protecting and enhancing local biodiversity through active management of the area, such as pruning, moving, planning and cleaning activities. They also actively monitor biodiversity in the area. Data shows a steady increase in biodiversity over time. Several red-list species have been spotted at the Ruige Hof, such as the bluethroat (Luscinia svecica), common kingfisher (Alcedo atthis) and bullfinch (Pyrrhula pyrrhula).

The Ruige Hof also focuses on co-benefits of biodiversity as outcome of their activities. Through education activities for adults and children, they hope to contribute to public support for and understanding of biodiversity conservation. In addition, they manage recreational facilities such as paths and benches. De Ruige Hof has about 450 members and over 50 active volunteers. Most of the annual budget of around €20.000 comes from membership contributions and donations.

Penllergare Valley Woods

Over the last 15 years, the Penllergare Valley Woods has been protected and restored by a local community group in Wales, UK (Penllergare, 2016). This community group, organised in the Penllergare Trust, owns and manages a 100 hectare landscape. It is a historic cultural landscape, consisting of a rich variety of trees, shrubs and exotic plants, two lakes and a waterfall which functions as a green corridor for a diverse range of wildlife. The trust was formed in 2000 by local residents as a (not for profit) company. It focuses on three purposes, in order of priority, the protection, conservation, restoration and maintenance of the landscape of Penllergare, promoting knowledge and appreciation of Penllergare and the protection and conservation of wildlife.

After previous developments, such as housing and truncation of the area by motorways, the trust was formed as response to ongoing development trends encroached upon the estate. Over 450 local volunteers, the financial support of sponsors and a Friends of Penllergare membership scheme contribute to maintain and to restore the Penllergare landscape and to halt further development of and adjacent to the area. Conservation focuses on the restoration of the lake, waterfalls and vegetation to improve habitats for wildlife, volunteers control invasive species, and replant former forested areas with broadleaf woodland. Volunteers report on and record sightings of interesting and locally rare wildlife. In addition, volunteer rangers provide education to children and adults, e.g. through wildflower walks.

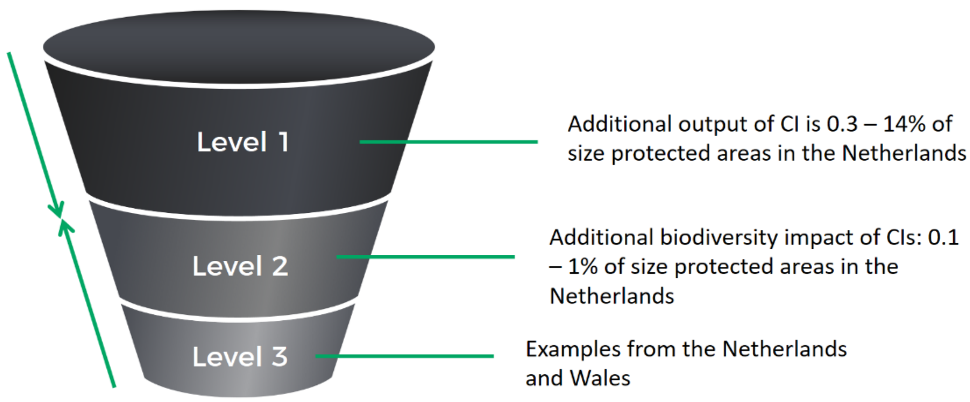


Figure 2.1 *Biodiversity outcome and impacts of CIs*

2.5. Expected future performance

Future performance depends on future demographics, socio-economic developments and trends in governance. In the UK and The Netherlands, the discourse on environmental governance has shifted in recent years, with an increases focus on non-state actors and a transformation of the state in parallel, including budget cuts (Buijs, Mattijssen, and Arts 2014; Ambrose-Oji et al. 2011). This supposed increase in CIs is in line with recent developments in other field, such as energy and personal care (Hajer et al. 2015). From a European perspective, future developments are hard to predict. Some argue that current trends towards devolution and self-governance in especially the Netherlands and the UK forebode similar trends in other North-West European countries (De Moor, 2016). Buizer et al (2015) identify several recent trends that may contribute to the rise and success of CIs across Europe. These include the emergence of new instruments for co-governance, including social media, participatory budgeting, crowd funding, and the right to challenge and the ongoing rise of urban agriculture and local food production. Nonetheless, evidence on whether the rise of self-governance in some North-West European societies forebodes a more general European trend is lacking.

2.6. Additionality to international biodiversity governance

CIs are characterised by citizens and grassroots initiating conservation efforts in addition to governmental effort. Indeed, they often emerge out of discontent with a lack of government efforts to protect natural areas against e.g. urbanisation (de Groot, Salverda, van Dam, et al. 2012; Mattijssen et al. Draft). The vast majority (approximately 80-95%) of CIs contributions relates to land formally not designated as Natura 2000 area (Mattijssen et al. Draft), which also indicates CIs contribution may be additional to state efforts. Although the contribution from CI may show some overlap with community forestry in especially North West European countries, most of it will be fully additional to government or NGO based contributions.

However, this additionality can also be questioned. It only holds for CIs autonomously developed outside government programs, not for initiatives initiated by governments as a kind of outsourcing. As such, current governmental focus in several countries on decentralisation, devolution and self-governance has been criticized as a form of neo-liberalisation (Uitermark 2015), characterised by retreating governments and a transfer of responsibility to the market and civil society. Such a retreat is then legitimised by the assumed rise in self-governance. From a critical perspective, the added value of CIs can thus be questioned. A trade-off may exist between a rise in CIs and a decrease in governmental efforts.

CIs may also relates to Community Forest Management. Consequently, overlap between the chapters on CIs and CFM may exist. Many examples of community forestry in the UK can both be considered a citizen initiative as well as examples of CFM. As the calculations in this chapter are predominantly based on quantitative studies in the Netherlands and CFM is still quite minimal in The Netherlands, such possible overlap is limited.

2.7. Conclusions

Citizens and local communities have a long history of engagements with and contributions to the conservation of biodiversity in a range of green areas. This involvement of citizens in protection, maintenance and enhancement of biodiversity is not primarily driven by government policies, but by intrinsic motivation of citizens. Usually CIs not only aim at benefits for biodiversity conservation, but also at co-benefits, such as developing accessible green spaces and at increasing environmental awareness. Because of the scattered and unorganised nature of CIs, only limited outcome and impact data are available. The current analysis is based on empirical results from only one country, the Netherlands (Mattijssen et al. 2016). The outcomes of the calculation are presented with high margins due to this high level of uncertainty.

In total, CIs contribute to nature management and restoration of 0.3% to 14% of the total designated areas in the Netherland. Estimated impact is significantly lower, between 0,1% and 1% of designated areas. Because of a lack of reliable data, possible impacts from political influence groups is not included in this calculation. Consequently, the actual contribution of CI's is expected to be significantly higher. The impact of green CIs is predominantly additional to efforts by governments or NGO's and are real impacts on the ground, not merely policy plans or bids that still need to be implemented. However, extrapolation from the Dutch context to a global context is difficult, because emergence and success of CIs depend strongly the environmental, social and governance context in each country.

While the main conclusion of this chapter may be that the direct impact of green CIs seem rather limited, indirect impact may outweigh these direct impacts. This includes pressure on state actors to improve biodiversity protection and a contribution to the implementation of governmental and NGO policies thought increased environmental awareness and public support for biodiversity conservation. Although these indirect impacts of CIs cannot be calculated, they seem relevant for all countries across the Globe.

3 'Community Forest Management' (CFM)

3.1. Introduction to community forest management

Community Forest Management (CFM) has become an influential approach in the management of forests around the world the last couple of decades (Agrawal, 2001; Arnold, 2001; Wiersum, 2009). Nearly 15% of these forests – about 500 million hectares – fall under such a management regime today (IFRI, 2015; RRI, 2014)[1]. As a response to state forestry and commercial timber production, and building upon traditions of customary regulations for forest commons, this approach puts fulfillment of local livelihoods and forest conservation first. In general, it can be defined as the use, management and conservation of forests by communities. Such forests may, may not, or may be partially owned by communities, and their management is often practiced in various degrees of collaboration with state forest agencies, donor organizations, knowledge institutions and/or companies. On one end of the extreme, forest management is fully community-based and the forests concerned are 100% owned by the community. Whereas on the other extreme communities just participate in some of the state forest management practices in public lands. Because of this variation, several terminologies are used to refer to these practices (community forestry, community-based forest management, community-managed forests, collaborative forest management, participatory forest management, joint forest management and forest co-management). We prefer the term CFM, because it is most referred to in the literature (based on a Google Scholar search).

The central idea behind Community Forest Management (CFM) is that local management of forests, either by communities or jointly with forest departments, is more effective than management by central state institutions, leading to both forests and people being better off. CFM brings 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 forest are regulated on paper, open-access regimes and tragedies of the commons remain in practice. In such cases, CFM can bring an attractive alternative for state forestry.

Already in the early 1970s, the idea of community participation, both for better forest management and for improving people's livelihoods, was practiced in a few countries, advocated by NGOs and scientists and intensively discussed in the FAO at global level (Arnold, 2001; FAO, 1978; Umans, 1993). Later, these ideas entered as norms into 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 early 1990s 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: (1) a conservation phase, in which CFM mainly targeted the conservation and rehabilitation of community forests; (2) an empowerment phase, in which the democratic and forest

1 This figure will be put into perspective in this Chapter below, even more so, will be lowered, actually. But this is what the current literature says.

rights of local communities were emphasized; (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 certification (Wiersum et al., 2013). Of course, these phases did not neatly follow up in time; rather, they have been overlapping and many aspects of these do still exist in parallel today.

3.2. Motives, goals, targets

The overall **motive** for CFM is to prevent deforestation and forest degradation, while improving forest-dependent livelihoods at the same time (De Jong, 2011). As such, it contrasts the classical forest conservation narrative of protected areas without people ('fortress nature') (Palomo et al., 2014). Such 'coercive conservation' led to exclusion of people from their lands and violation of their forest rights in many Tropical countries, 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).

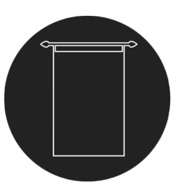
Overall, CFM exhibits two **goals**: (1) To enhance the sustainable management of community forests; and (2) To improve forest-related livelihoods for local people. Sometimes, a third goal is also strived for: (3) To empower the community vis-a-vis the state. In this case study, we focus on the first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

CFM does not include global **targets**. The ones that come closest are some of those under the Aichi Targets of the CBD, particularly target 2 (full integration of biodiversity values in development and poverty reduction strategies by 2020), target 5 (rate of forest loss and rate of habitat loss halved in 2020 and, if feasible, close to zero), target 7 (making agriculture, fishery and forestry fully sustainable by 2020), target 11 (increase of protected areas up to 17% of terrestrial lands in 2020, including co-management of national parks), target 14 (restoring and safeguarding ecosystem services relevant for water, health and livelihoods) and target 18 (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, and halt and reverse land degradation and halt biodiversity loss).

3.3. Theory of change: from input to impact

The theory of change varies with the type of CFM. Let's take the two extremes from the above. Firstly, in case 'formal ownership' of a forest adjacent to a community is transferred from the state to a community, the theory says that 'ownership' is crucial for change (Ostrom et al., 2002). Once owned by oneself, people will feel much more responsible for the resource, so that deforestation and forest degradation will be substantially reduced, or even halted. The idea is that you will not destroy your own property, but that you will try your best to manage it as sustainably as possible. And once forest management is properly planned on paper and forest use is practiced in a sustainable way along the lines of such a plan, both people and trees will profit (and, indirectly, forest biodiversity too). In a scheme:


Input: Ownership


Output: Forest management plans

Outcome: Sustainable forest management practices

Impact: improved livelihoods and forest condition (including biodiversity)

Secondly, in case management responsibilities are shared between the state and the community in a state-owned forest, the crucial factor is not 'ownership', but 'awareness raising' and 'exchange' (Arts and Babili, 2013). The state forest department (or an NGO) makes communities aware that public forests need to be properly managed in peoples' own interest, because if not, they will lose the resource in the (near) future, and the products and services derived from it as well. Once this vision is shared, communities are granted certain management responsibilities and financial resources from the state in exchange of abstaining from certain adverse forest use practices. In a scheme:

Input: awareness raising

Output: Sharing of responsibilities and resources

Outcome: Sustainable forest management practices

Impact: improved livelihoods and forest condition (including biodiversity)

In this case study, we include both theories of change, because we are mainly interested in outcomes and impacts, and these are the same for both (sustainable forest management practices as outcome and improved livelihoods and forest condition as impact).

3.4. Past performance: assessing outcome and impact

We follow the so-called 'funnel-shaped assessment methodology', as outlined in Chapter 1 of this report (see *Figure 1.5*, page 5). Level 1 refers to general data on number of initiatives or hectares of managed lands and/or protected nature by ICIBs. The second level is much more specific in that it tries to identify proportional data on hectares, initiatives or nature that show 'real' impact, or in other word, actually contribute to the enhancement, protection and/or enrichment of biodiversity. The 3rd level is even more specific: it deals with some detailed examples from the 2nd level sample, to show how positive biodiversity impact looks like on the ground.

Box 3.1.: Assessing CFM impact: steps taken

Level 1

Step 1: Assessment of the global forest area owned and/or managed by indigenous peoples and local communities, to be summarized as 'forest commons', based on two datasets (FAO, 2015; RRI, 2014).

Step 2: Assessment of how much of these global forest commons fall under the 'new' CFM regime, including forest biodiversity conservation objectives, based on trend analysis (FAO, 2015; RRI, 2014).

Level 2

Step 3: Assessment of the proportion of CFM cases that contributes positively to forest biodiversity conservation, based on three meta studies (so-called 'benchmark'; Arts and De Koning, 2017; Bowler et al. 2012; Persha et al., 2011).

Step 4: Assessment of the global forest area under CFM that contributes positively to forest biodiversity conservation (based on steps 1-3).

Step 5: Assessment of the average improvement of forest biodiversity under CFM, based on MSA analysis (Schipper et al., 2016; MSA stands for 'Mean Species Abundance').

Level 3

Step 6: Illustration of positive forest biodiversity impact through CFM in two specific cases (Arts and De Koning, 2017).

Other

Step 7: Scenario analysis for 2020 and 2030, based on trend analysis (FAO, 2015; RRI, 2014) and on assumptions related to two dynamic parameters: (a) growth of forest area under CFM over time and (b) growth of proportionality of CFM area with positive forest biodiversity impact. Three scenarios will be presented (business as usual, high-high scenario and a low-low scenario).

Step 8: Additionality of CFM area with positive forest biodiversity impact, compared to formally protected areas (PAs) and formally forest protected areas (FPAs) in the world (based on steps 4 and 7).

Step 9: Assessment of degree of overlap of: (a) CFM as non-state initiative (=ICIB) and as state initiative (=non-ICIB); and (b) case studies in this report (particularly this CFM case with forest certification and forest restoration).

Besides following the 'funnel', this chapter adds some analysis on future scenarios, additionality and overlap. Box 1 above summarizes all steps taken below.

CFM: Level 1

For this 1st level, we work with two datasets, one from the FAO Global Forest Assessment (FAO-GFA; http://www.fao.org/forest-resources-assessment/ explore-data/flude/en/), and one from the Rights & Resources Initiative (RRI; http://rightsandresources.org/en/resources/ tenure-data/tenure-data-tool/). Both include data on global forest area, changes in global forest area over time and forest tenure and management arrangements, including data on indigenous peoples and local communities. *Table 3.1.* gives an overview of these datasets, which indicates that both are different in terms of years of assessment and in terms of numbers on forest areas and tenure. Moreover, both sets are incomplete.

Table 3.1 *Overview of rough data and omissions*

	Global forest cover (FAO-GFA, 1980-2015) *Billions of hectares*	Global forest area owned and/or managed by indigenous peoples and local communities (FAO-GFA, 2015) *Millions of hectares*	Global forest cover (RRI, 2014) *Billions of hectares*	Global forest area owned and/or managed by indigenous peoples and local communities (RRI, 2014) *Millions of hectares*
1980	2.1*	-	-	-
1990	3.4	93.3	-	-
2000	3.9	169.5	-	-
2002	-	-	3.6	385.3
2010	4.0	400.0	-	-
2013	-	-	3.3	512.7
2015	4.0	-	-	-
2020	-	-	-	-
***Tropical developing countries only**				

From the table above, we learn that - according to FAO data - indigenous peoples and local communities owned and/or managed 400 million hectares of forests in 2010 (which amounts to about 10% of global forest cover, as indicated by FAO-GFA), whereas the data of RRI show that three years later, in 2013, indigenous peoples and local communities owned and/or managed 513 million hectares of forests around the world (which amounts to about 15% of global forest cover, as indicated by RRI). These figures are not compatible, because these are based on different sources and approaches. Whereas FAO-GFA data are based on incomplete country statistics, delivered voluntary by governments around the world, RRI data are based on 52 countries and 73% of the world's forests and extrapolated to global level. Moreover, RRI data are delivered by experts, not by state bureaucracies. As a consequence, RRI data probably overestimate the share of forests managed and/or owned by indigenous peoples and local communities, because these are extrapolated from 52 most forested countries in the world, whereas FAO data probably underestimate this share, because its data set is incomplete. Moreover, the latter is particularly contested. For example Hansen et al. 2013 calculate much higher global deforestation rates for the period 2000-2012 (-12.5 million hectares / year, which is about two to three time higher than FAO figures). Yet their dataset does not predate the year 2000 and does not include data on forest land tenure and management. Such makes Hansen's dataset less relevant for our analysis. All in all, we take the FAO-GFA and RRI datasets as indicating a valid range of forest areas managed and/or owned by indigenous peoples and local communities. So about 400 to 515 million hectares, which equals about 10 to 15% of the world's forests, are to be considered 'forest commons'.

It is tempting to take these figures as forest commons being managed according to CFM approaches and principles. However, not all these forests in the world are under the 'new' CFM regime for sustainable forest management and forest biodiversity conservation. Some forest commons are an heritage of the past, others have been recently installed or re-introduced to improve forest management through either decentralized, community-based forest management or through joint, participatory forest management, respectively. We assume that the 'old' forest commons are still used or managed according to business-as-usual practices. This may be or may not be positive for biodiversity, but biodiversity objectives were definitely not explicitly strived for before the 1980s. Therefore, to be at the safe side of the equation, we decided to exclude the 'old' forest commons from our calculations below. As a threshold, we take 1980 as a starting point. CFM started in the 1970s (Arnold, 2001), but really took off from the 1980s onwards. Hence, those forest commons that we still observe in 1980 are assumed to be the 'old' ones, not influenced by the 'new' CFM sustainability and biodiversity paradigm.

To distinguish between these two, we need 'time series data' on global forest area owned and/or managed by indigenous peoples and local communities. We calculate time series of both datasets FAO-GFA and RRI from *Table 3.1.* Starting point is 1980, but we also include the near future, up to 2030, in order to be able to assess the expected future performances of CFM in section 5 below. The FAO-GFA dataset delivers CFM area information from 1990, 2000 and 2010 (see *Table 3.1.* in the above), from which we can deduce that global forest area owned and/or managed by indigenous peoples and local communities increased with 7.6 million hectares annually in the 1990s and with 23.1 million hectares annually in the 2000s. If we assume that CFM expansion continued in the same degree after 2000, but grew exponentially in the beginning of CFM in the period 1980-2000, as is generally claimed in the scholarly literature (Arnold, 2001), we can deduce annual CFM expansion figures per decade as expressed in *Table 3.2.* For RRI data, we however followed a different logic, because we only have data available for two points in time (2002 and 2013; see table 3.1). We now assume that CFM expansion occurred evenly over all decades. Since both datasets are incompatible (see the discussion in the above), we do not think these two different assumptions about CFM expansion – exponential initial growth for the FAO-GFA dataset but even growth for the RRI dataset over all decades – is problematic. Again, both datasets are different anyway, and together they show us a range of data on CFM.

Table 3.2 *Annual expansion of forest owned and/or managed by indigenous people and local communities per dataset and per decade*

	Annual expansion (FAO-GFA) million Ha.	Annual expansion (RRI) million Ha.
1980s	2.5	11.6
1990s	**7.6**	11.6
2000s	**23.1**	**11.6**
2010s	23.1	11.6
2020s	23.1	11.6

Bold figures *are 'real' ones, adopted from Table 3.1; the other ones are deductions based upon certain assumptions about CFM expansion over time.*

Table 3.3 below is based on information from Tables 3.1 and 3.2. It shows time series data on global CFM area in millions of hectares per decade of both datasets. It also includes extrapolations for 2015, 2020 and 2030 in order to distract expectations on future CFM performance in the next section.

***Table 3.3** Time series from the FAO-GFA and RRI datasets on forest area owned and/or managed by indigenous peoples and local communities; calculations are based on figures from Tables 3.1 and 3.2.*

	Global forest area owned and/or managed by indigenous peoples and local communities (FAO-GFA, 2015) Millions of hectares	Forest area owned and/or managed by indigenous peoples and local communities (RRI, 2014) Millions of hectares
1980	68.3	130.0
1990	**93.3**	246.0
2000	**169.5**	362.0
2002	-	**385.2**
2010	**400.0**	478.0
2013	-	**512.7**
2015	516.0	535.9
2020	631.0	593.9
2030	862.0	709.9
***Bold figures** are 'real' ones, adopted from Table 1; the other ones are deductions based upon certain assumptions about CFM expansion over time.*		

From Table 3.3., we can now calculate the forest commons under the 'new' CFM regime. For FAO-GFA, this amounts to 331.7 million hectares (400 – 68.3 million hectares, see Table 3); for RRI, the size of this area is 382.7 million hectares (512.7 – 130 million hectares, see Table 3). Therefore we conclude that global forest area under CFM, including forest biodiversity conservation objectives, amounts to about 330 – 380 million hectares.

CFM: Level 2

Of course, not all forests under the 'new' CFM regime do actually contribute to forest biodiversity conservation on the ground, since we know from numerous field studies from all over the world that the results of CFM policies, programmes and projects are generally mixed (Charnley and Poe, 2007). In other words, there exist 'implementation deficits' all over the place. To distil a proportion from the above figures that indeed shows positive biodiversity (-related)[2] impacts, we use three multi-N reviews of CFM research: (1) Arts and De Koning, 2017; (2) Bowler et al. 2012; and (3) Persha et al., 2011. The first source synthesizes the CFM database of the Forest and Nature Conservation Policy Group (FNP) of Wageningen University; the second one produces a systemic literature review of CFM papers in 12 electronic databases (including Scopus and Web of Sciences) and the third one synthesizes research of the International Forestry and Resources Institutions (IFRI) network. All three reviews include CFM cases from all over the world and time series data of CFM initiatives, while two of them privilege (quasi)experiments over other methodologies to assess the impacts of CFM (Bowler et al.; Persha et al.). Table 3.4. presents an overview of the three reviews.

From table 3.4 on the next page we observe that about 80% of CFM cases improve forest conditions, like increase of forest area, increase of basal area (of tree stems at breast height) or of tree density (8 of 10 cases

2 The concept of 'diversity' refers to the number of species and their distribution in a system. The more they are evenly distributed in a system, the higher the diversity. Many CFM studies, though, use 'species richness' as an (more simple) indicator, just referring to the number of species. And to make things even more complex, below we will also use the concept of 'species abundance', an indicator which includes number of species and number of individuals in a system. Ideally, all indicators referring to 'biodiversity' in this study would have been similar. But they are not, given that we apply secondary analyses. So if we speak of 'positive biodiversity impact' below, the indicators behind this statement can be different, and therefore one should actually read 'positive biodiversity-related impact'.

Table 3.4 *Biodiversity impacts of CFM initiatives*

	N CFM studies in the review	Positive forest condition outcomes (basal) area and tree density)	Positive biodiversity impact (species richness, high-value forests)
Arts & De Koning	10	8	4
Bowler et al.	42	34	-
Persha et al.	84	-	31

in Arts & De Koning, 2015; and 34 of 42 cases in Bowler et al., 2012). But such does not necessarily imply biodiversity impact. For example, planting new trees (often commercial species) and managing standing trees (for more timber harvest and non-timber forest products) can definitely increase (basal) area and tree density, but positive effects on forest biodiversity are often negligible. Yet, case studies do show that CFM can produce substantial positive biodiversity impacts on the ground as well. Enrichment planting, reintroduction of tree species and wildlife, their natural return due to better forest conditions and the protection of high-value forests from deforestation and degradation are examples of how CFM performs. From the table we learn that about 35-40% of CFM cases show such positive biodiversity (-related) impact (4 of 10 cases in Arts & De Koning, 2015; and 31 of 84 cases in Persha et al. 2011). To be on the safe side, though, and given the much bigger sample of Persha et al., we stick to the lower benchmark of 35%.

From tables 3.3. and 3.4. we can now calculate the proportion of CFM area that exhibits positive biodiversity impact for both datasets. For FAO-GFA, we have to multiply 0.35*331.7 million hectares, which amounts to 116 million hectares in 2010. For RRI, we have to multiply 0.35*382.7 million hectares and this amounts to 134 million hectares in 2013. Since these datasets are not compatible, as argued before, we use both to sketch a range of hectares of global CFM territory that produces positive forest biodiversity impacts: about 115 to 135 million hectares in 2010-13 (rounded up to 5 million).

Besides numbers of hectares with positive biodiversity impact, it is also interesting to know the degree of such positive impact. Here, like in Chapter 3 on forest restoration and in Chapter 6 on voluntary sustainability standards, we use MSA ('Mean Species Abundance') as a biodiversity indicator, a method developed by PBL (w.globio.info; Schppers et al., 2016). MSA is based both on the number of species and on the abundance of individual species; and it includes various taxonomic groups (plants, mammals, birds, reptiles, invertebrates, etc.). The indicator gives an estimate of the 'naturalness' of for example a forest, by estimating the degree to which a disturbed or managed forest resembles a natural situation. 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. This means that about 70% of the species populations that are usually present in natural forests can still be found in lightly used forests. Table 3.5. shows the MSA of more and less managed and disturbed forests.

Table 3.5 *MSAlu valuges assigned to GLOBIO land-use classes. Sources: Alkemade et al. 2009; Alkemade et al. 2013; GLOBIO reference database (www.globio.info).*

Globio land use class	MSA_{LU}
Forest - Natural	1.0
Forest - Plantation	0.30
Forest - Clear-cut harvesting	0.50
Forest - Selective logging	0.70
Forest - Reduced impact logging	.085

We now use these estimates to assess how CFM might positively affect biodiversity. To do so, we distinguish five trajectories of forest conservation and sustainable use under CFM (see Table 6): from degraded forest to plantation[3] , from plantation to managed secondary forest, from the latter to semi-natural managed forest, from the latter to semi-natural forest with reduced impact logging (RIL) and a final trajectory of 'avoided loss', which means that CFM prevents a conversion of the forest to a non-forested land use type[4]. In the last trajectory, the state of a forest (of whatever type) is maintained against deforestation pressure. Each trajectory produces a certain positive MSA effect over time (see Table 6). Since we do not know the total global forest area under each of these trajectories, we assume that they are all equally present in current CFM forests with positive biodiversity impact around the world. Based on this assumption, we estimate that on average 'mean species abundance' (MSA) increases with 25% under CFM conditions that allow for positive biodiversity impact. Of course, such increase in MSA is not immediately realized once an CFM regime is installed. Based on knowledge of various cases (Arts and De Koning, 2017), this might take between 10 to 30 years to materialize (depending on type of trajectory and type of forest).

Table 3.6 *MSA gains through various CFM trajectories*

State of forests before CFM	MSA.	State of forests after CFM	MSA	MSA Effect
Degraded + exploited	0.1	Plantation forestry	0.3	+0.2
Plantation forestry	0.3	Managed, secondary forest	0.5	+0.2
Managed, secondary forest	0.5	Managed, semi-natural forest	0.7	+0.2
Managed semi-natural forest	0.7	Semi-natural forests, RIL	0.85	+0.15
Any state (excl degraded)	0.3-0.85	Same state ('avoided loss')	0.3-0.85	+0.3-0.85
Average (in case all trajectories of forest conservation and sustainable use and of avoided loss are equally divided over global CFM territory)				+0.25

CFM: Level 3

Here, we present two examples of CFM with positive biodiversity impact from the FNP dataset (Arts and De Koning, 2017). See boxes 1 and 2 below.

Box 3.2.

The study of De Koning (2011) addresses CFM in a Bolivian community in the Amazon region. It looked at how a community collectively managed their forest resources. As the communal forest area was already substantial in size and rich in biodiversity, the new CFM regime did not lead to an increase in forest size, or an improvement of forest conditions. But it particularly led to 'avoided deforestation' in an high-value, biodiversity-rich area, as it offered the community a way to enhance their livelihoods from standing forests and trees, in particular the collection of Brazil nuts and the selective harvest of valuable timber. Moreover, land titles of the communal forest areas were formalized through CFM, providing the community the necessary stability in access to forest resources, additional income and even to medical services (in which part of the forest revenue was collectively invested). Whereas similar forests were clear-cut for pastures and crops in Bolivia, this forest area under CFM has remained standing.

3 We acknowledge that we argued otherwise in the case of plantation forestry in the setting of the positive biodiversity impact benchmark of 35% in the text above. There, we argued that tree plantation forestry does not produce substantial positive biodiversity impact. But the MSA methodology of PBL argues that it does. Being consistent would then imply that the benchmark increases to 80% (see Table 4 in the above). Intuitively, we consider such a benchmark far too high, given all stories of failure in the CFM literature. On the other hand, 35% might be too low (see next footnote). Final remark: biodiversity and MSA are different concepts. The latter includes species abundance, so the absolute number of individuals, the former not. This also explains the different 'diversity perspectives' on plantation forestry.

4 Here, another discrepancy with the setting of the 35% benchmark surfaces. In our secondary analysis of Persha et al., we did not include 'avoided loss' (so cases with zero increase of tree species richness). If we had done so, another 6 cases would have been added to the calculation, increasing the 'biodiversity success rate' from 37 tot 44% in this meta-analysis. Together with the one of Arts and De Koning, a benchmark of 40% is then more justified. In that case, the area of CFM with positive biodiversity impact would have increased to 130-155 million Ha. However, we decided to keep the numbers as they are in the main text, because we favour a cautious assessment over an (too) optimistic one.

Box 3.3.

This case study, reported in Arts and Babili (2013), is about an forest area under CFM in four North-Tanzanian neighbouring villages. Forest conditions and biodiversity impacts were assessed through satellite images, focus groups, interviews and field observations over time. Results showed the following. About 85 per cent of the respondents had observed an improvement in forest conditions and status of biodiversity since the introduction of CFM in the 1990s. They mentioned an increase of the forest cover, more reliable water springs, growth of more grasses (fodder!), reintroduction of some lost tree species (for example African teak), less soil erosion on forested slopes, and an increase of wildlife, particularly monkeys and leopards (received with mixed feelings by the villagers). Some of these perceptions, particularly related to the change in forest cover, could be validated by GIS data. A time series of satellite images of the four village forests under CFM – about 2,800 hectares in total – revealed an increase in forest cover over time. Whereas this cover declined by about 50 hectares in the 1990s, there was a gain of about 100 hectares in the first decade of the 21st century.

CFM outcome and impact: conclusion

Figure 3.1 below summarizes the conclusions of the above analysis. Although CFM has never been designed as a mechanism for the conservation of biodiversity per se – like protected areas – it nonetheless produces positive biodiversity impacts through sustainable land use and management practices. Of course, this is not achieved in all CFM initiatives. Of about the 330-385 million hectares of forests which fall under CFM regimes today, about 35% performs well in terms of positive biodiversity impact. This amounts to 115-135 million hectares of forests. In those forests, 'naturalness' (MSA) increases with 25% on average. Examples from Bolivia and Tanzania show how the trajectories of avoided loss, expansion of forests and enrichment of forests work, with concrete examples of social and biodiversity benefits (granting of land titles, new or return of tree species, increase of wildlife, increase of timber and non-timber forest products, enhancement of water protection, decrease of erosion, etc.).

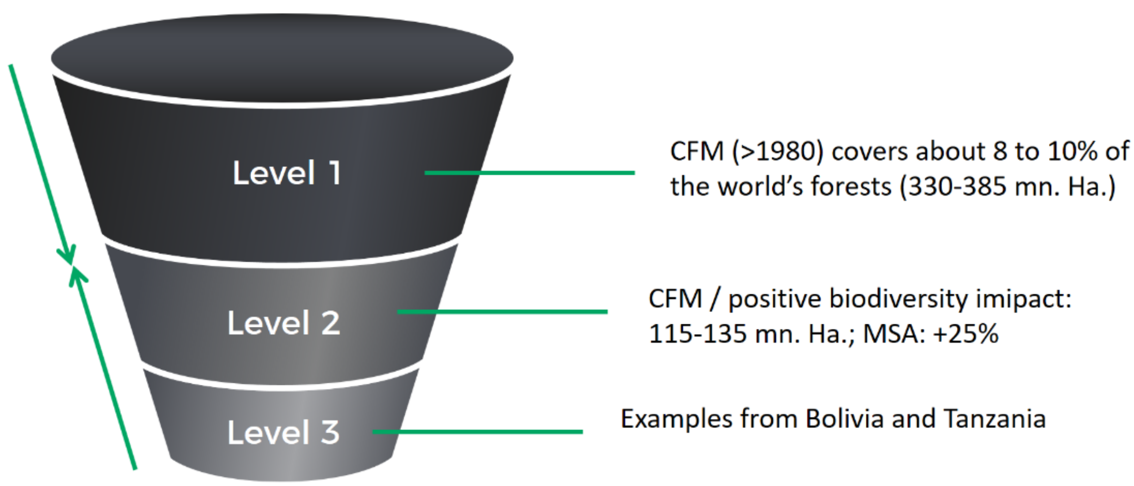


***Figure 3.1** Biodiversity outcomes and impacts of CFM*

3.5. Expected Future Performance

Table 3.7 below shows the figures of CFM areas with positive biodiversity impact in millions of hectares valid for both datasets of the recent past, currently and in the near future. These figures are deduced from Table 3. Total forest commons are firstly converted into CFM areas under the 'new' regime of sustainable use and biodiversity conservation (size of CFM area of respective future dates minus the size from 1980) and secondly converted into CFM areas with positive biodiversity impact (using the 35% benchmark). The outcomes are presented in Table 3.7. and we call these figures the 'business as usual' (BAU) scenario, since past trends are simply extrapolated to the future. Such implies that ranges of global CFM area that positively contribute to biodiversity conservation in the future under a BUA scenario are the following (rounded up to 5 million of hectares): 140-155 million hectares in 2015; 160-195 million hectares in 2020; and 205-280 million hectares in 2030.

Table 3.7 *CFM area with positive biodiversity impact in the recent past, currently and in the near future; deduced from Table 4; 'business-as-usual' (BUA) scenario*

	FAO-GFA dataset (2015) *Millions of hectares*	**RRI dataset (2014)** *Millions of hectares*
2010-13	115	135
2015	157	142
2020	197	162
2030	278	203

Besides this BUA scenario, we distinguish two other ones based upon two dimensions: (1) growth of gross area of community forestry (with 23.1 million hectares of annual growth of CFM area under the FAO-GFA dataset and 11.6 million hectares of annual growth of CFM area under the RRI dataset as 'business-as-usual' benchmarks); and (2) growth of proportionality of positive biodiversity impact in global forest commons under CFM (with 35% proportionality benchmark as 'business as usual'). Based on those two dimensions, we can distinguish a high-high (HH) and a low-low (LL) scenario[5]. For HH we assume a doubling of CFM area growth and 50% proportionality of positive biodiversity impact since 2015; and for LL a halving of area growth and 20% proportionality since 2015, respectively. In box 4, the HH and LL calculations are presented, whereas Tables 8 and 9 present the outcomes of these two scenario.

Box 3.4.

HH scenario steps:

1. *Subtract Ha. 2015 from Ha. 2020 under BAU scenario (see Table 7)*
2. *Double the number (increased area growth)*
3. *Multiply outcome with 0.5/0.35 (increased growth of proportionality)*
4. *Add result of 3 to Ha. 2015 to calculate Ha. 2020 under HH scenario*
5. *Extrapolate trend 2015-2020 to 2020-2030 to calculate Ha. 2030 under HH scenario*

LL scenario steps:

1. *Subtract Ha. 2015 from Ha. 2020 under BAU scenario (see Table 7)*
2. *Halve the number (decreased area growth)*
3. *Multiply outcome with 0.2/0.35 (decreased growth of proportionality)*
4. *Add result of 3 to Ha. 2015 to calculate Ha. 2020 under LL scenario*
5. *Extrapolate trend 2015-2020 to 2020-2030 to calculate Ha. 2030 under LL scenario*

Table 3.8 *HH Scenario*

	FAO-GFA dataset (2015) *Millions of hectares*	**RRI dataset (2014)** *Millions of hectares*
2015	157	142
2020	271	199
2030	499	313

5 In theory, one can also add a high-low (HL) and a low-high (LH) scenarios, but since we are only interested in possible ranges of future CFM area figures, and not so much in various scenario narratives and policy recommendations, we limit ourselves to the two extremes HH and LL

Table 3.9 *LL Scenario*

	FAO-GFA dataset (2015) *Millions of hectares*	**RRI dataset (2014)** *Millions of hectares*
2015	157	142
2020	168	148
2030	190	160

From these LL, BUA and HH scenarios, we can infer the range of global CFM area with positive biodiversity impact for 2020 and 2030. In case we: (1) round up all numbers to 5 million hectares; (2) take the highest numbers from the HH scenario; (3) the lowest ones from the LL scenario; and (4) the average ones from the BAU scenario, the figures as presented in Table 10 emerge. These show that **future biodiversity performance of CFM ranges between 150 and 270 million hectares in 2020 and between 160 and 500 million hectares in 2030.**

Table 3.10 *Range of future CFM area with positive biodiversity impact under different scenarios (million hectares)*

Scenario	LL	BAU	HH
2020	150	180	270
2030	160	240	500

Of course, a fourth, 'doom' scenario might exist: a collapse of the CFM movement in combination with strong re-nationalization of forest policies under future governments. In such a scenario, the global area under CFM will probably decrease and the positive impacts on biodiversity in the area which is still left will go down as well, due to demotivation of communities, NGOs, forest departments, etc. Then, concrete figures in 2020 and 2030 will probably drop below the 2010-13 range of CFM performance (115-135 million hectares). And the average increase of 'naturalness' of forests under CFM (about 25% increase of MSA) will drop as well. However, we provisionally assume that such a 'doom' scenario is not very likely to occur, at least not soon, since the movement is still growing and forest decentralization and devolution initiatives are still increasing in numbers.

3.6. Additionality to international biodiversity governance

Although CFM was not initiated for biodiversity conservation per se, it does nonetheless contribute to it, as the above makes clear. An interesting question is therefore how much it 'adds' to classical nature protection. In order to assess this, we need to compare the contribution of CFM with the performance of nature conservation in general. Currently, about 15% of the Earth's terrestrial surface is set aside for protected areas (GBO IV, 2014), which amounts to about 2,248 million hectares[6]. If we compare this figure with the surface of CFM that currently contributes to biodiversity conservation – 115-135 million hectares – we can conclude that its additionality ranges between 5 and 6%. If we however restrict this comparison to Forest Protected Areas (FPAs) around the world (about 650 million hectares in total; Morales-Hidalgo, 2015), additionality increases to 18-21%. Thus, CFM substantially contributes to achieving both Aichi and SDG targets (see section 2).

So far we assume that no overlap exists between CFM areas on the one hand and FPAs on the other, but probably there is. For example, extractive reserves in the Amazon can have both a protective and an CFM

6 See: http://hypertextbook.com/facts/2001/DanielChen.shtml. Several Earth surfaces, including land and water, have been calculated by various research institutes on the basis of various methodologies and assumptions, including NASA; this internet source shows five of these, from which the average is taken.

status (Nepstad et al., 2006). Unfortunately, we do not have the data to quantify this overlap. A similar issue of overlap exists among the case studies in this report, particularly among CFM, forest certification and forest restoration. For example, about 6 million hectares of community managed forests are currently certified by FSC (oral communication of FSC staff); that amounts to an overlap of about 5% (compared to CFM area that positively contributes to biodiversity conservation). Concerning overlap of CFM with forest restoration programs, data are unfortunately unknown. Finally, we should take overlap among non-state and state actors' initiatives in ICIBs into account. After all, many CFM initiatives have been hybrid in nature, with contributions of governments, international organizations, NGOs, scientists and communities alike. In addition, states have been in the lead in CFM in some countries (India, Nepal), whereas non-state actors have been so in others (Tanzania, Ethiopia). Again, this overlap is difficult to quantify.


Habitat
Restoration
Area

4 Landscape Restoration

4.1. Introduction to landscape restoration

Restoration of landscapes by indigenous communities has been taking place for hundreds and thousands of years (Bhakta et al. 2016). 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 programmes like in China. In the last few decades there has been increasing international attention for landscape restoration, heavily driven by non-state actors such as:

- The Landscapes for People, Food and Nature Initiative (LPFN). Launched by EcoAgriculture partners in November 2011, LPFN is an international collaboration for knowledge sharing, dialogue and action on integrated landscape management in order to achieve improved food production, ecosystem conservation, and sustainable livelihoods.

- The Global Partnership on Forest and Landscape Restoration (GPFLR). The GPFLR was launched in 2003 by IUCN, WWF and the Forestry Commission of Great Britain to drive and support a network of governments, international and non-governmental organizations with diverse efforts to restore degraded forests and lands that deliver benefits to local communities and to nature (Forestlandscaperestoration.org, 27Sept2016).

- The Global Restoration Initiative (consisting of WRI, IUCN and other partners) works with governments and international partners to inspire, enable and implement restoration on degraded landscapes. WRI has identified more than 2 billion ha of cleared and degraded forest and agricultural land suitable for restoration (roughly twice the size of China). With this data, the Global Restoration Initiative (GRI) aims to accelerate restoration of degraded land into sustainable agriculture, agroforestry and forested landscapes. Partners of the GRI are the the Governors' Climate and Forests Task Force (GCF Task Force), GPFLR and LPFN (see above).

Landscapes restoration is highly relevant in biodiversity governance, as agriculture and deforestation are the largest drivers for biodiversity loss. Restoration activities often restore or rehabilitate lands, but at the same time might halt and/or prevent further degradation and conserve the landscape as it is. Restoration efforts can focus on different land use systems, such as forests, degraded lands, agricultural areas and natural, pristine ecosystems. The added value of landscape restoration for biodiversity is the integrated approach towards ecosystems. Restoring ecosystems and its functions with a landscape approach has the aim to improve human well-being it offers a holistic and realistic approach for multi-functional landscapes, in which conservation and ecological restoration efforts are in balance with other land uses, including sustainable agriculture and agroforestry to support economic development (GLF, 270916).

In recent years, landscape restoration has been included in ambitious cross-sectoral national targets and gained a more prominent position on the international agenda. Both country level commitments and the Bonn Challenge pledges show evidence of the increased political will for restoration. However, much uncertainty remains on the extent to which these commitments will be implemented and what the actual impact of these efforts will be (Wentink, 2015).

The Bonn Challenge was launched in 2011 by the GPFLR, an International Cooperative Initiative (ICI), with the aim of being a practical means of realizing existing international climate change, biodiversity and land degradation commitments through restoration. It is a global effort to restore 150 million hectares of the world's deforested and degraded land by 2020 (later extended to 350 million hectares by 2030; Bonn Challenge website). In this analysis, we zoom in on the Bonn Challenge pledges to look at the possible impact on biodiversity. This analysis has a strong link with the other case-studies on Voluntary Sustainability Standards and community forestry initiatives, as a large part of restoration is focused on forests.

Box 4.1. Landscape restoration as a concept

The scientific field of "restoration ecology" emerged in the 1980s, soon followed by the foundation of the Society for Ecological Restoration (SER) in 1987. Hence, being an ecological concept initially, restoration is often interpreted as returning an ecosystem 'back' to a natural, historical, pristine ecosystem (also: ecological restoration). Other concepts related to restoration refer to practices like the cleaning of polluted ecosystems (environmental restoration) or bringing unproductive, 'destroyed' land, like former mining sites, back into production (rehabilitation).

In more recent decades, with issues coming up on the political and scientific agenda such as food security, water(shed) management, climate change adaptation, there was a growing need for a more integrated approach towards ecosystems and the efforts to restore them. Thus, the concept of landscape restoration came on the agenda that surpasses the older interpretation of restoration as it aims to restore ecological integrity as well as improve human well-being through multifunctional landscapes. Landscape restoration addresses a wide 'landscape' of stakeholders, technologies and ecosystem services.

4.2. Motives, goals, targets

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

Goals: Landscape restoration has the objective to restore ecological integrity as well as improve human well-being through multifunctional landscapes. The Bonn Challenge specifically aims to offer a practical means of realizing existing international climate change, biodiversity and land degradation commitments such as SDG 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'.

Targets: Landscape restoration offers an integrated way to achieve multiple (inter)nationally agreed goals. It contributes to Aichi targets 5 'Rate of forest loss and rate of habitat loss halved in 2020 and, if feasible, close to zero', target 14 'Ecosystems that provide essential services and contribute to health, livelihoods and well-being, are restored and safeguarded by 2020', and target 15 'ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration by 2020'. SDG 15 integrates these Aichi targets and more, and includes, similar to the landscape approach, a wider concept of restoration that incorporates also prevention of further loss, the conservation of what we have and inclusion of human wellbeing in restoration strategies. By 2020…

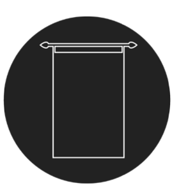
- 15.1 ..ensure the conservation, restoration and sustainable use of terrestrial and inland freshwater ecosystems and their services (…).
- 15.2 ..promote the implementation of sustainable management of all types of forests, halt deforestation, restore degraded forests and substantially increase afforestation and reforestation globally
- 15.3 ..combat desertification, restore degraded land and soil, including land affected by desertification, drought and floods, and strive to achieve a land degradation-neutral world
- This target specifically originates from UNCCDs Land degradation neutrality goal shaped at UN Conference on Sustainable Development (Rio+20) where world leaders agreed "to strive to achieve a land-degradation neutral world".
- 15.4 (By 2030) ..ensure the conservation of mountain ecosystems, including their biodiversity, in order to enhance their capacity to provide benefits that are essential for sustainable development
- 15.5 ..take urgent and significant action to reduce the degradation of natural habitats, halt the loss of biodiversity and protect and prevent the extinction of threatened species
- 15.6 ..promote fair and equitable sharing of the benefits arising from the utilization of genetic resources and promote appropriate access to such resources, as internationally agreed
- 15.7 ..take urgent action to end poaching and trafficking of protected species of flora and fauna and address both demand and supply of illegal wildlife products
- 15.8 ..introduce measures to prevent the introduction and significantly reduce the impact of invasive alien species on land and water ecosystems and control or eradicate the priority species
- 15.9 ..integrate ecosystem and biodiversity values into national and local planning, development processes, poverty reduction strategies and accounts

The Bonn Challenge pledges are to support existing international objectives, such as REDD+ (reducing emissions from deforestation and forest degradation, UNFCCC), which is a framework through which developing countries are rewarded financially for any emissions reductions achieved through decreasing and halting deforestation and forest degradation, by conserving and enhancing forest carbon stocks, and by the sustainable management of forests.

4.3. Theory of Change: from input to impact

While the Bonn challenge could be considered an output of the GPFLR, it is here considered as a starting point to analyse its (possible) impacts. We apply an input-output-outcome-impact framework to describe the theory of change.

Input: the Bonn Challenge pledge was initiated by the GPFLR, an international cooperative of non-state organisations and governments. The input of the Bonn Challenge provides is to instigate the restoration of 150 million ha of degraded and deforested lands by 2020 (and 350 million ha by 2030). Initially, the idea behind Bonn Challenge was to provide a platform where both state and non-state actors could pledge their commitments for realizing such goals. But in practice, Bonn Challenge pledges were mainly signed by national governments. As such, the input of non-state actors may be found in the implementation of these state pledges.



Output: since the Bonn Challenge launch in 2011, a variety of pledges have been made, followed by new or adjusted domestic targets and policies, and projects and programs on the ground for landscape restoration. These policies and programs usually focus on restoration and/or conservation of a certain ecosystem function (biodiversity, water, soil fertility, carbon storage etc.), with secondary effects on other ecosystem functions, and can be done by both state and non-state actors, through national programs or local community initiatives supported by NGOs.

Outcome: landscape restoration will be visible in the implementation of a wide variety of restoration practices on the ground, such as the adoption of sustainable land use technologies (e.g. terraces, vegetative barriers), reforestation (e.g. plantations or natural regeneration), agroforestry and silvopastures by land owners and managers (e.g. farmers, foresters, NGOs, local communities). As such, landscape restoration is a collection of efforts, with a variety of goals. These goals are not always intuitive (i.e. contrary to our intuition, natural forest fires should not always be prevented but controlled, and planting more trees is not always the most sustainable solution), and can change over time. Landscape restoration projects may incorporate one of the abovementioned activities in the entire landscape, but more often combine several within a mosaic landscape, a mixed landscape, or the inclusion of for instance ecological corridors in production lands (pastures, agriculture, plantations).

Indicators showing potential outcomes of the Bonn Challenge are the amount of hectares on which landscape restoration practices are (planned to be) implemented. Unfortunately, actual adoption of these practices is still too early to determine, as the Bonn Challenge is a relative new and still ongoing process, of which little reporting and monitoring of on-the-ground implementation has been done.

Impact: the actual impact of the Bonn Challenge on biodiversity depends on the specific ecosystem function improvement that a restoration effort aims for. For instance the increase in biodiversity when a smallholder farmer shifts his/her practice to agroforestry is smaller compared to the increase in biodiversity when reforestation through natural regeneration takes place on abandoned, degraded land. The level 1, 2, 3 funnel analysis in the next chapter will provide an indication of the actual impact the Bonn Challenge has on biodiversity, in quantities of improves Mean Species Abundance (MSA %) as indicator for biodiversity and area in hectares (ha). However, as the Bonn Challenge is it itself more of a political instigator than a monitoring instrument, and is still an ongoing process up to 2020 and 2030, the impact on biodiversity that the challenge will eventually have cannot be determined yet. Also, the contribution of International Cooperative Initiatives on Biodiversity (ICIBs) to this impact is therefore yet unknown and is in many ways indirect.

Box 4.2. China case study as example for the dynamics between outcome and impact

Often, the analysis following the i-o-o-i framework will show that in practice impacts will be achieved in an iterative process with loops going back and forth between stages. An example of such dynamics can be found in the case of landscape restoration in the Loess Plateau in China.

China managed to restore huge amounts of land in the last 40 years (good outcomes). However, the high rate of restoration was achieved using minimally diverse or single species plantings, resulting in low seedling survival rates (low impact). Also, many local communities were unable to benefit from the restored forests. These results drew increasing demand to ‘re-green’ China in a more natural way.

For example, take the “Conversion of Cropland to Forests Program” (CCFP), also known as “Grain-for-Green” or the “Sloping Land Conversion Program,”, launched in 1999 by the Chinese government (output). Reviews, especially of the first pilot phase of the programme, also note low seedling survival rates and monoculture-reforestation (Trac et al. 2013) (low impact). However, more recent reviews (covering phase 2 of the programme) mention higher survival rates (Gutiérrez Rodríguez et al. 2016), although the success rate vary per region and depend on how the practices are implemented with local communities (Bennett el at 2014) (2nd loop, improved impact). This learning process is now ready for third loop in CCFP phase 3 (high impact).

Box 4.3. Assessing impact of Landscape restoration in the Bonn Challenge: methods used

0) General context: Calculations are based on the restoration commitments pledges under the Bonn Challenge, domestic restoration plans of countries where data is available (from the Bonn Challenge website), restoration potential as calculated by GPFLR and impact on biodiversity with Mean Species Abundance as indicator.

1) Level 1: Calculating overall number of initiatives / hectares etc.
Step 1a: GPFLR restoration opportunities, estimated through the global extent of degraded land available for restoration (Bonn Challenge website): for mosaic restoration, wide-scale restoration and unpopulated lands.
Step 1b: Facts of the Bonn Challenge: 1) target up to 2020; 2) target up to 2030; 3) pledges up tillpresent day, in amount of commitments and amount of hectares.
Step 1c: The total amount of domestic targets for restoring degraded and deforested lands.
Step 1d: calculate what the Bonn Challenge (targets and pledges) and domestic targets represent in the total restoration potential of step 1a.

2) Level 2: Calculating proportion of initiatives/hectares with biodiversity impact
Step 2a: With Mean Species Abundance (MSA) as biodiversity indicator, we look at MSA values per land use, as found in Schippers et al 2016.
Step 2b: As no data is available of the planned restoration efforts and following land use changes under the Bonn Challenge itself, we use restoration activities of the domestic target plans as indicator for that. We sum up the total amount of hectares in the restoration classes used in those domestic targets, and calculate the change in MSA using step 2a for the restoration classes.
Step 2c: With the change in MSA per restoration activity and the total sum of hectares per class, we calculate the total expected impact on biodiversity in MSA of these domestic targets. And calculate that back to the potential biodiversity impact of the Bonn Challenge.

3) Level 3: Examples of impact from level 2 sample
Step 3: more fine-tuned calculations and detailed description of how restoration might actually be implemented and how CI might play a role in that, we look at two Bonn Challenge examples: the Mata Atlantica Restoration Pact in Brazil (and with that indirect effects on Brazil's pledge as well) and the Democratic Republic of the Congo.

4.4. Past performance: assessing outcome and impact

Whereas level 1 from the funnel mentioned in our methodology is quite straightforward, as it represents the total amount of ha pledged to be restored under the Bonn Challenge as percentage of total restoration opportunity map, this is not the case for level 2 analysis. As the Bonn Challenge is relatively 'young' and its goals have yet to be realised by 2020 and 2030, policy design is still ongoing in many countries, let alone that implementation has started. Hardly any reporting mechanisms have been put in place to monitor progress, and consequently few data are available on the type of restoration efforts of the Bonn Challenge in total that are needed for the level 2 impact analysis. Therefore, the level 2 analysis will determine the possible impact of Bonn Challenge pledges on biodiversity based on expected land use changes and changing Mean Species Abundance (MSA) values as a result of that. Subsequently, level 3 will provide detailed case examples of two of the largest pledges, Brazil and Democratic Republic of the Congo (DRC).

Level 1: global numbers on LR commitments (in ha or %)

The GPFLR found restoration opportunities with the potential to improve both ecosystems and human wellbeing everywhere, but especially in tropical and temperate areas. They estimated the global extent of degraded land available for restoration to be (Bonn Challenge website):

- 1.5 billion ha suitable for mosaic restoration, in which forests are combined with other land uses, including agroforestry, small-holder agriculture, and buffer plantings around settlements.
- About 0.5 billion ha suitable for wide-scale restoration, i.e. closed and continuous forests.
- 0.2 ha of unpopulated lands, mainly in the far northern boreal forests, that have been degraded by fire, suitable for passive restoration, e.g. natural regeneration.
- This totals to about 2.2 billion hectare of land that might be suitable for restoration.

The Bonn Challenge is to have 150 million ha of land pledged to be restored in 2020, and 350 million in 2030. This is 6.82% and 15.91% respectively, of the 2.2 billion ha restoration opportunity.

To date, 136.32 million hectare is pledged to be restored in 39 different commitments (Jan. 16, 2017). This is 6.20% of the 2.2 billion ha restoration opportunity.

Outside of the Bonn Challenge, many countries have existing domestic targets for restoring degraded and deforested lands, found in official government plans, REDD+ strategies, and in-country multilateral investment programs. Through these plans, landscape restoration will turn from plan to action. Currently, these domestic targets sum up to 199.36 million ha, would amount to 9.06% of the 2.2 billion ha of land suitable for restoration (*Annex 1*).

Thus, to sum up, in percentage of the estimates restoration opportunity, this means:

- Bonn Challenge goal 150 million ha by 2020 ⟶ 6.82%
- Bonn Challenge goal 350 million ha by 2030 ⟶ 15.91%
- Bonn Challenge actual commitments 136.32 million ha ⟶ 6.20%
- Domestic targets (includes both 2020 and 2030 targets) ⟶ 9.06%

This does not mean these lands will be restored by 2020, but there is a political commitment to start doing so. In that sense, the actual impact that the pledged 136.32 million ha has on biodiversity will depend on: translation of this pledge into actual programmes and projects, the enabling environment for these programmes to be successfully implemented, and the type of restoration efforts that will be done on the ground in view of desired function improvement.

The Bonn Challenge is thus a first step in what can possibly be restored. However, the 350 million ha to be restored under the challenge in 2030 offer a way, albeit 10 years later, to reach CBDs Aichi Target 15: "By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks have been enhanced, (…) including restoration of at least 15% of degraded ecosystems, (…)".

Level 2: amount of actual programmes and projects to be implemented

MSA changes for land use changes

MSA changes are based on Alkemade et al. 2009 (table below) and the land use changes defined in the domestic targets (*Annex 2*).

Table 4.1 *MSA values for Land cover classes - Adapted from Alkemade et al. 2009*

Main GLC class*	Sub-category	Description	MSALU
Snow and ice	Primary vegetation	Areas permanently covered with snow or ice considered as undisturbed areas	1.0
Bare areas	Primary vegetation	Areas permanently without vegetation (e.g. deserts, high alpine areas)	1.0
Forests (1, 2, 3, 4, 5, 6, 7, 8, 9, 10)	Primary vegetation (forest)	Minimal disturbance, where flora and fauna species abundance are near pristine	1.0
	Lightly used natural forest	Forests with extractive use and associated disturbance like hunting and selective logging, where timber extraction is followed by a long period of re-growth with naturally occurring tree species	0.7
	Secondary forests	Areas originally covered with forest or woodlands, where vegetation has been removed, forest is re-growing or has a different cover and is no longer in use	0.5
	Forest plantation	Planted forest often with exotic species	0.2
Scrublands and grasslands (11, 12, 13, 14, 15)	Primary vegetation (grass- or scrublands)		1.0
	Livestock grazing	Grasslands where wildlife is replaced by grazing livestock	0.7
	Man-made pastures	Forests and woodlands that have been converted to grasslands for livestock grazing.	0.1
Mosaic: cropland/forest	Agroforestry	Agricultural production intercropped with (native) trees. Trees are kept for shade or as wind shelter	0.5

Main GLC class*	Sub-category	Description	MSALU
Cultivated and managed areas (16, 17)	Low-input agriculture	Subsistence and traditional farming, extensive farming, and low external input agriculture	0.3
	Intensive agriculture	High external input agriculture, conventional agriculture, mostly with a degree of regional specialization, irrigation-based agriculture, drainage-based agriculture.	0.1
Artificial surfaces	Built-up areas	Areas more than 80% built up	0.05
* 1, Broad-leaved evergreen forest; 2, closed broad-leaved deciduous forest; 3, open broad-leaved deciduous forest; 4, evergreen needle-leafed forest; 5, deciduous needle-leaved forest; 6, mixed forest; 7, swamp forest; 8, mangrove and other saline swamps; 9, mosaic: forest/other natural vegetation; 10, burnt forest; 11, evergreen scrubland; 12, deciduous scrubland; 13, grassland; 14, sparse scrubland and grassland; 15, flooded grassland and scrubland; 16, cultivated and managed areas; 17, mosaic: cropland/other natural vegetation			

Box 4.4. Categories of restoration efforts under domestic targets, as defined on the Bonn Challenge website

Forest land (Land where forest is, or is planned to become, the dominant land use)
- If the land is without trees, there are two options:
  - ***Planted forests*** and woodlots (Planting of trees on formerly forested land. Native species or exotics and for various purposes, fuel- wood, timber, building, poles, fruit production, etc.)
  - ***Natural regeneration*** (Natural regeneration of formerly forested land. Often the site is highly degraded and no longer able to fulfill its past function – e.g. agriculture. If the site is heavily degraded and no longer has seed sources, some planting will probably be required.)
- If the land is degraded forests:
  - ***Silviculture*** (Enhancement of existing forests and woodlands of diminished quality and stocking, e.g., by reducing fire and grazing and by liberation thinning, enrichment planting, etc.)

Agricultural land (Land which is being managed to produce food)
- If the land is under permanent management:
  - Agroforestry (Establishment and management of trees on active agricultural land (under shifting agriculture), either through planting or regeneration, to improve crop productivity, provide dry season fodder, increase soil fertility, enhance water retention, etc.)
- If it is under intermittent management:
  - **Improved fallow** (Establishment and management of trees on fallow agricultural land to improve productivity, e.g. through fire control, extending the fallow period, etc., with the knowledge and intention that eventually this land will revert back to active agriculture.)

Protective land and buffers (Land that is vulnerable to, or critical in safeguarding against, catastrophic events)
- If degraded mangrove:
  - **Mangrove restoration** (Establishment or enhancement of mangroves along coastal areas and in estuaries.)
- If other protective land or buffer:
  - **Watershed protection and erosion control** (Establishment and enhancement of forests on very steep sloping land, along water courses, in areas that naturally flood and around critical water bodies.)

To calculate the MSA increases resulting from the restoration activities as categorized by the Bonn Challenge website (listed below, and see box no 4.4 above. see box above), with the MSA values as found in *table 4.1, we f*ind:

- **Planted forests and woodlots**: from 0.1 (former forest, pasture or abandoned land) to 0.2 (exotic species) or 0.5 MSA (native species). Increase is 0.1-0.4 MSA, so average of 0.25 MSA increase
- **Natural regeneration**: from 0.1 (former forest, pasture or abandoned land) to 0.5 MSA (native species) or 0.7 (secondary forest). Increase is 0.4-0.6 MSA, so average of 0.5 MSA increase.
- **Silviculture**: from 0.5 (degraded forests) to 0.7 (lightly used forests). Increase is 0.2 MSA.
- **Agroforestry**: from 0.1- 0.3 (intensive or small-holder agriculture) to 0.5 MSA (agroforestry). Increase is 0.2-0.4 MSA, so an average of 0.3 MSA increase.
- **Improved fallow**: from 0.3 (recent fallow land, we assume is comparable to extensive agriculture) to 0.5 MSA (we assume improved fallow is similar to secondary forests or agroforestry in terms of MSA). Increase is 0.2 MSA.
- **Mangrove restoration**: from 0.5 (secondary forests) to 0.7 MSA (lightly used forests). Increase of 0.2 MSA.
- **Watershed protection and erosion control**: as trees here are just added as protective measure, their main goal here is not to increase biodiversity. We estimate thus a mere 0.1 MSA increase.

Table 4.2 *Domestic targets: Land use changes in ha (See Annex 2)*

Restoration categorie	Amount of ha	Percentage of total are
Planted forests and woodlots	103,719,067 (+438,700)	52.02%
Natural regeneration	14,499,107	7.27%
Silviculture	43,833,844 (+200,000)	21.99%
Agroforestry	29,869,512 (+220,000)	14.98%
Improved fallow	600,000	0.30%
Mangrove restoration	388,554	0.19%
Watershed protection and erosion control	6,454,750	3.24%
Total	199,364,834 (+858,700)	100%
Note that the (+xxxx) numbers derive from ranges in the source data (e.g. 100 – 350, would then give 100 (+250)). Thus the first number represents the minimum of that range, the addition gives the maximum. We choose to use the minimum values to continue calculations here, to be conservative, as all of these plans are still under implementation and ambitious as they are, these goals are to reach as it is. As for the average potential impact on biodiversity, leaving out these top range values represent <1% of the total goals anyway, and their influence is therefore minor.		

Thus on average the domestic targets… (52.02% with +0.25 MSA, 7.27% with +0.5 MSA, 21.99% with +0.2 MSA, 14.98% with +0.3 MSA, 0.30% with +0.2 MSA, 0.19% with +0.2 MSA and 3.24% with +0.1 MSA) …give 199.36 million ha with an average 0.26 MSA increase.

Level 3: From action to successful action

In level 2, we used the domestic targets as proxy for the Bonn Challenge impact. Here, under level 3, we will zoom into the 12 million ha pledged by Brazil and some examples of the influence that Cooperative Initiatives (CIs) may have on the Bonn Challenge pledges.

The remaining 10 million ha add to the regional Initiative 20x20, through Brazil's Low-Carbon Agriculture Plan (ABC Plan), with 5 million ha of integrated crop, livestock and forest management and 5 million ha of recovering degraded pastures (Biderman et al. 2016).

If we look into the NDC (Nationally Determined Contribution) to implement the 12 million ha Bonn Challenge pledge of Brazil, it states the intention of “restoring and reforesting 12 million hectares of forests by 2030, for multiple purposes”. This 12 million will be implemented through the Law to Protect Native Vegetation, known as the Forest Code. The implementation of the Forest Code is described in the National Plan for the Recovery of Native Vegetation (PLANAVEG) (MMA, 2014).

The PLANAVEG estimates that there is a deficit of 21 million hectares all Brazilian biogeographical regions that could be restored (MMA, 2014). Of this 21 million ha, …

- …16.4 million hectares are in Legal Reserves (RL). Maximum amount of RL that could be “offset” by Environmental Reserve Quotas (CRA) is about 56%, or 9.2 million hectares. In addition, 5 million hectares of land need to be restored to Conservation Unit (UC). 30% of these have private property rights over them. The purchase of these 1.5 million hectares could be financed by land owners with RL deficits, thus offsetting their deficits. As a result, the actual amount to be restored is 5.7 million ha in Legal Reserves (RL) (16.4 - 9.2 - 1.5). This is the lower limit of the range because some land owners with the potential to generate and sell CRA may choose not to make use of this mechanism million ha to be restored (MMA, 2014).
- …4.6 million ha is to be restored in Permanent Preservation Areas (APP).
- In addition to this 10.3 million ha (RL + APP), some recovery actions on degraded lands with low productivity are likely to be made to improve ownership and diversify incomes through new business flows and revenues, promote recreation and leisure, among other reasons. These can account for over 2 million ha additional to the goal of the National Plan.
- The above sums up to 12,3 million ha to meet the Bonn Challenge (MMA, 2014).

In terms of actual impact on biodiversity (from the average MSA values and total hectares above), are expected to be:

- 5.7 million ha of RL:
  - o Legal reserves contain native vegetation cover but can be restored using up to 50 percent of exotic species in the beginning of the restoration activities (Kissinger, 2014).
- MSA increase comparable to ‘planted forest’ (see level 2[1]): increase of 0.25 MSA
  - o 4.6 million ha of APP:
- APPs: “those covered by native vegetation, currently protected or not, including riparian areas, areas surrounding lakes, mangroves and areas containing steep slopes and that are important sources of natural capital” (Kissinger, 2014)
  - o MSA increase either 0 if already protected, or 0.4 (from secondary or lightly used forest to (almost) pristine forest), but also comparable to mangrove restoration (0.2 MSA increase, see level 2). Note that for simplification we will use an average 0.2 MSA increase.
- 2 million ha of additional efforts on degraded lands and improved land practices: this may contain agroforestry (+0.3 MSA), improved fallow (+0.2 MSA) and watershed protection and erosion control (+0.1 MSA). So we use an average of 0.2 MSA increase for these hectares as well.

The potential impact on biodiversity of Brazil’s Bonn Challenge pledge is therefore an average MSA increase of 0.22.

The role of CIs on the implementation of the Bonn Challenge
Example: Brazil’s Atlantic Forest (Mata Atlântica) Restoration Pact

ICIBs play an important role in Brazilian landscape restoration. The Atlantic Forest (Mata Atlântica) Restoration Pact (AFRP), launched in 2009 by a Public Private Partnership (PPP) of NGOs, private companies,

1 see level 2, page .., on how the MSA values for Bonn Challenge categories, e.g. ‘planted forest’, were derived.

local governments, and research institutions, has over 260 members. The AFRP pledged to restore 1 million ha of Atlantic Forest under the Bonn Challenge by 2020 (and 15 million ha of forest by 2050), before the Brazilian government followed with a 12 million ha pledge in Brazil altogether.

The AFRP aims to promote biodiversity conservation, job creation and income generating opportunities through the restoration supply chain, provision of key ecosystem services to millions of people as well as to establish incentives for landowners to comply with the Revised Brazilian Forest Code (2012). This law requires landowners to protect and/or restore Areas of Permanent Protection (APPs) and Legal Reserves (LR) within their properties and to spatially identify and register them in the Environmental Rural Registry (CAR). Part of the efforts of the AFRP is to measure actual implementation and implement a monitoring system for the CAR. Up to December 31, 2016, more than 3.92 million rural properties have been registered in CAR, totaling an area of 399.23 million ha (http://www.florestal.gov.br; accessed 1012017).

At present only 7-20% of the original Atlantic Forest (Mata Atlântica) area is left, due to a history of extensive deforestation. And although it is too early to report on impact as their work is still ongoing, the AFRP First Evaluation Report mentions that the AFRP coordinates and manages over 80 different projects, representing almost 60.000 ha under restoration (Melo et al., 2013). These results are promising, also in terms of actual impact on biodiversity. Not only does the AFRP already aim to restore 1 million ha by 2020 (8,3% of Brazils total Bonn Challenge pledge), but most of their efforts, such as setting up a monitoring system for the CAR, have a positive effect elsewhere in Brazil as well.

Example: CI as watchdog in the Democratic Republic of the Congo

The success of the Brazilian Atlantic Forest Restoration Pact, however, is more exception than rule, in that CIs play a crucial role in the design, monitoring and implementation of the restoration efforts instigated by their own Bonn Challenge Pledge. On the other end of this spectrum, we find the Democratic Republic of the Congo (DRC).

The pledge by Democratic Republic of the Congo (DR Congo) to restore 8 million ha is part of the African Restoration Initiative (AFR100) (Bonn Challenge website; 21Dec2016). Their pledge is incorporated in plans and initiatives of the DRC government, and REDD+ activities will be a big part of meeting their pledge. The REDD+ mechanism was launched in DRC in 2009, followed by an ambitious Readiness plan by DRC in 2010. However, the actual implementation of these REDD+ plans is under threat by other activities and competition from other government policies, and both REDD+ pilot projects struggled to get started, planned studies remained largely incomplete, and could therefore not provide the input needed for the national strategy. This delay in actual implementation is visible when looking at the actual impact on forest area as well, with forest area decreasing about -0,2 mln ha between 2010-2015, at a similar rate to previous years (www.fao.org/forestry/fra/fra2010/en/).

In order to adhere to the schedule for producing the national strategy by 1 January 2013, the country went developed a national REDD+ strategy framework that was presented at COP18 in Doha in December 2012. It marks the start of a second phase of the REDD process, the investment phase from 2013 to 2016 (Kipalu et al. 2013).

The role of CIs in DRC in relation to meeting the pledge is therefore not in the role of active implementing stakeholder, but rather to serve as 'watchdog'. Take for instance, the collaborative mapping initiative Moabi, which aims to increase transparency and accountability on resource issues in DRC. Moabi is part of a multi-partner project to create institutions and tools for independently monitoring of natural resources in DRC. The project's current focus is developing an independent monitoring approach for REDD+ (http://rdc.moabi.org/the-initiative/en/).

Figure 4.2 below summarizes the conclusions of the level 1,2,3 analysis. Other than ecological restoration and protected areas, landscape restoration under the Bonn Challenges was not specifically designed for the conservation of biodiversity. Nonetheless, it does have a positive impact on biodiversity, both as direct effect or by-product of restoration practices and landscape approaches implemented in the landscapes that are being restored under the Bonn Challenge.

As the Bonn Challenge's first milestone is in 2020, many countries are still in the process of designing National Action plans. However, the case in Brazil does indeed prove that ICIB's might serve as crucial factor in going from the level of legislation and National Programmes to actual successful implementation by covering crucial enabling conditions needed for on-the-ground action and sustainable conservation and protection.

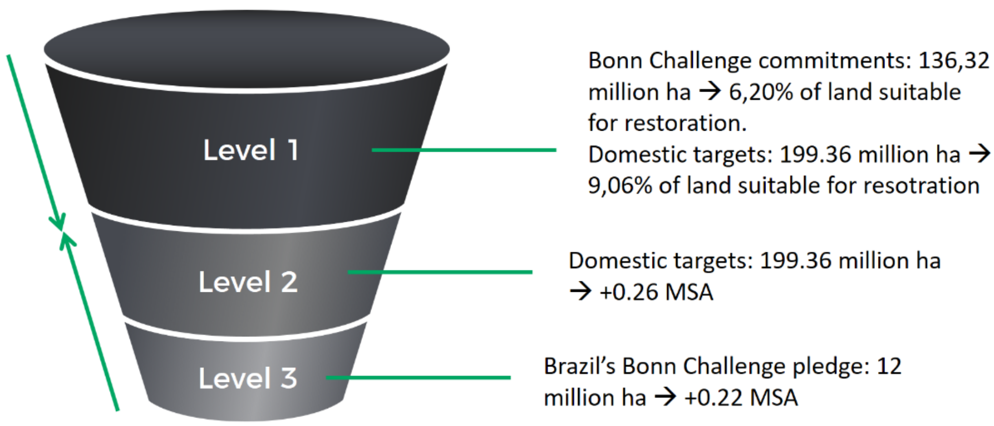


Figure 4.2 *Biodiversity outcome of landscape and forest restoration*

The Bonn Challenge as an example of International Cooperative Initiatives (ICIs) on Biodiversity is a bit of an odd one out, in the sense that although it started and was founded with the idea of involving non-state actors in such pledges, reality at present is that most pledges were actually done by state actors (e.g. national governments). This is why, in the examples of level 3 we have tried to indicate in which way ICIBs specifically have and can address the impact of these (state) pledges. Also, when addressing the concept of 'impact' in this case study, this includes the impact on biodiversity and ecosystem services in balance with the wellbeing of land users. However, it is important to take into account that (although extremely relevant for local livelihoods) the issues around land rights were not within the scope of this study.

Note as well that what is not taken in account are possible displacement effects of restoration efforts, and the positive or negative indirect effects of measures outside of the areas where they are implemented.
And finally, as the Bonn Challenge case focusses especially on land(scape) restoration pledges, the pledges itself often include efforts of rehabilitation and conservation measures that prevent further loss and halt ongoing degradation processes. In such, the definition of restoration in this study is relatively wide.

4.5. Expected Future Performance

In the coming years, the Bonn Challenge pledges have to be developed into actual regional, sub-regional and national action plans and programmes, and from project design to actual implementation. For this, several regional implementation platforms for the Bonn Challenge have emerged, such ministerial roundtables in Latin America, East and Central Africa, and Asia-Pacific (bonnchallenge.org, 27Sept2016) and:

- Initiative 20x20 is a country-led effort in Latin America and the Caribbean region to restore 20 million ha of land by 2020, with US$730 million of private investment. Seven Latin American and Caribbean countries and two regional programs have committed to begin restoring 27.7 million hectares (roughly the size of the UK) of degraded land by 2020. WRI, CATIE, CIAT, and IUCN support Initiative 20x20 through their Global Restoration Initiative (WRI website 261016).

- The Governors' Climate and Forests Task Force (GCF Task Force) is a subnational collaboration between 29 states and provinces from Brazil, Indonesia, Ivory Coast, Mexico, Nigeria, Peru, Spain, and the United States. Originally focused on jurisdictional programs such as reduced emissions from deforestation and land use (REDD+), the GCF Task Force have now put forest and landscape restoration on their agenda as well.
- The AFR100 (African Restoration Initiative) will support two types of restoration activities:
  - Mosaic landscapes: trees on agricultural land (planting or natural regeneration), through agroforestry (trees & crops) and silvopasture (trees & livestock).
  - Forest restoration (planting or natural regeneration) on degraded / deforested land
  - Commitments announced through AFR100 support the Bonn Challenge, the New York Declaration on Forests and the African Resilient Landscapes Initiative (ARLI).

The success of this implementation will depend on various enabling conditions. Based on several case studies, the following conditions stand out: political momentum, safeguarding restoration quality, trade-offs are acknowledged and addressed, stakeholder involvement on different levels, multi-sector involvement, supporting regulations & legislation, financial incentives, and available and accessible information. See *Annex 3* for an elaboration on this list.

At the moment, there are 39 commitments under the Bonn Challenge, in 33 countries, mostly from African and Central and Latin American countries (see *figure 4.3* below). The list of domestic targets is similar, but includes more Asian as well. But it is not hard to imagine that in addition to the possible impact of the current pledges, there is much more potential for restoration should new countries join the Bonn Challenge and start designing and implementing their own domestic programs as well. It is especially high income regions, including Europe, Japan and Australia that are still absent from the Bonn Challenge list of commitments. Although this does not necessarily mean they do not have restoration programs, there is added value in committing through a pledge as well as to be able to give out a message of ambition, support political momentum for restoration and work together to share experiences and on monitoring the actual implementation of targets and ambitions on the ground.

Figure 4.3 *From Bonn Challenge website: the Bonn Challenge commitments (Feb. 2017)*

4.6. Additionality to international biodiversity governance

To understand the context of the Bonn Challenge it is important to note that many countries already had existing plans and targets for restoring degraded and deforested lands, for which the Bonn Challenge pledges are more of supporting political tool. These can be found in official government plans, REDD+ strategies, and in-country multilateral investment programs (IUCN accumulated the domestic targets per country; Bonn Challenge website). These numbers are shown in the last column of *Annex 1*. It is interesting to see the different roles the Bonn Challenge might play:

- For quite a number of countries, the amount of hectares under the Bonn Challenge pledge is larger than Domestic Targets. Here, the Bonn Challenge could function as both a support for existing, and encouragement for additional domestic targets.
- For a smaller number of countries, the amount of hectares under the Bonn Challenge pledge is smaller than Domestic Targets. Here, the Bonn Challenge is mostly a support for (part of) the existing targets. In Brazil, for instance, the Bonn Challenge is implemented through domestic plans for reforestation, restoration and protection of forest. Grassland and other ecosystems are restored in targets outside of the challenge. Especially in these cases, additionally with the Community Forest Management (CFM) and Voluntary Sustainability Standards (VSS) cases in this report is very likely.
- Quite a few countries have committed under the Bonn Challenge but have no domestic targets to support that. These countries are still at the input stage of the i-o-o-i framework. Here, the Bonn Challenge is an indicator, or might have been an important instigator, for for increased political momentum for restoration.
- There is also a set of countries however, especially in Asia, that have domestic targets already but no Bonn Challenge pledge. The pledge here plays a less vital role for restoration, and might serve to receive more recognition or additional political support for existing restoration efforts.

5 Renaturing Cities

5.1. Introduction to renaturing cities

Currently over 50% of the world's 7.4 billion people live in cities. It is expected that by 2050, with an estimated population of over 9 billion people, 70% will live in cities. As cities adapt to accommodate an increasing population urban expansion is inevitable. According to the Cities and Biodiversity Outlook in 2012 over half of the projected urban area in 2050 was yet to be built. This urban expansion, or sprawl, puts an ever-increasing stress on natural resources, land and ecosystem services (CBD 2012). Many cities are situated in biodiversity rich areas such as floodplains (Elmqvist 2013). As rapid urban expansion occurs it leads to habitat loss and may have severe consequences for biodiversity, especially in cities bordering on biodiversity hotspots (CBD 2012).

As evermore cities become aware of the sustainability challenges they face, more and more cities adopt sustainability policies which benefit urban biodiversity. These policies may range from integrated sustainability strategies to dedicated biodiversity action plans. In this context renaturing cities is presented as an agenda or strategy to address urban biodiversity, mainly through maximising ecosystem services provided by urban green infrastructure (Connop et al. 2016). This case study identifies cities that have dedicated themselves to urban biodiversity conservation, either implicitly or explicitly through a renaturing cities agenda.

A wide variety of actors may contribute to urban biodiversity conservation, regardless of whether they are formally involved in a renaturing cities agenda. This study will focus on policy initiatives by local governments and international non-state partnerships. Local governments play an important role in renaturing cities strategies, both in terms policy as well as implementation. City governments, municipalities and districts can act within the powers delegated to them by national government. This allows for complementary activities relative to local governments to complement national policy (UNCBD 2010: X/22). On the international level there are a number of non-state organizations which mostly act as facilitators. Such organizations include ICLEI – Local Governments for Sustainability, C40 Cities, Eurocities, and URBIS initiatives.

The study provides a rough estimate of the relative number of cities involved in renaturing cities and urban biodiversity policies. Moreover, it identifies ways forward to determine and quantify the effect these policies have on urban biodiversity. Quantification should ultimately allow for an analysis of how non-state initiatives benefit biodiversity as compared to state-led initiatives. Finally, this study provides an estimate for the future relative number of cities involved in renaturing cities and urban biodiversity policies.

5.2. Motives, goals, targets

Motives

The motives for a renaturing cities agenda or an urban biodiversity policy varies between cities, but the main driver is often very similar; expected benefit from the improvement of ecosystem services. Actors rely on the benefits which urban ecosystems deliver. The framework by Connop et al. (2016) depicts the city as a social-ecological system with the four categories of ecosystem services being present. (I) Provisional services include urban agriculture and fresh water; (II) regulating services include air quality and carbon sequestration; (III) cultural services include education and human well-being; and (IV) habitat services, including biodiversity. Urban biodiversity policies generally include a variety of these services, and initiatives often support the social-ecological system as a whole.

Goals

The theoretical goal of renaturing cities is to maximise ecosystem services provided by urban green infrastructure (Connop et al. 2016). Urban green infrastructures (UGI) are interconnected networks of green spaces. The wide variety of ecosystem services they provide range from conservation of biodiversity and facilitating human health and well-being to climate change adaptation (Hansen and Lorance Rall 2014). There is a wide variety of green spaces that can contribute to urban green infrastructure. The GREEN SURGE project provides a typology, Report Milestone 23 (Haase et al. 2015) lists 40 different components, including amongst others, parks, residential areas, building greens and urban agriculture.

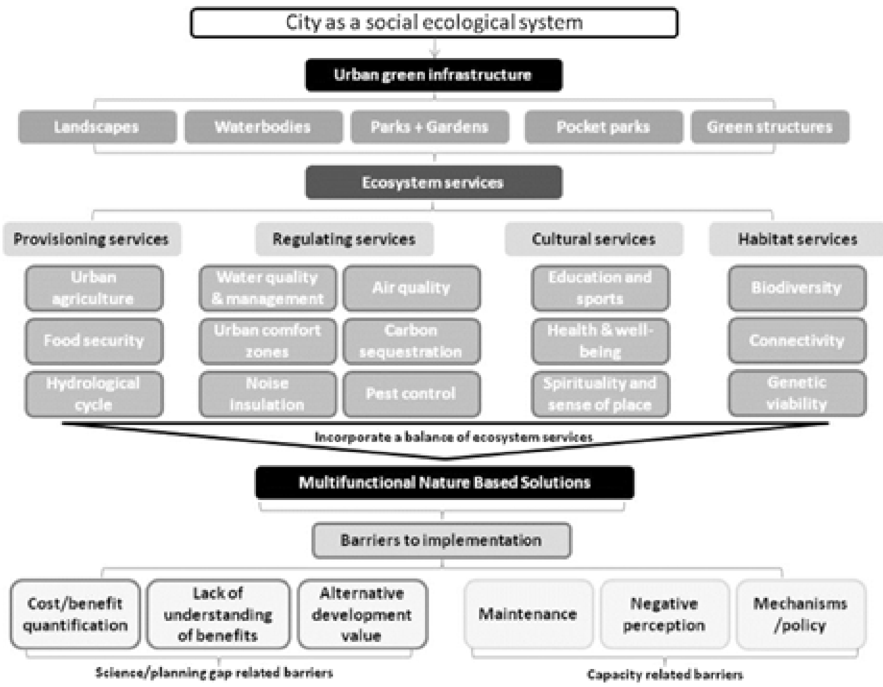


Figure 5.1 *City as a social ecological system (Connop et al. 2016)*

In practice, goals for renaturing cities depend on each initiative. Relevant to this study are the goals relating to ecosystem services, outlined Local Biodiversity Strategy and Action Plans (LBSAP) or similar documents. Specifically, those goals which directly relate to biodiversity are of interest. Such goals include "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).

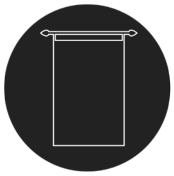
Targets

Of the plans identified and reviewed in this study, very few set out detailed targets. ICLEI's Durban Commitment encourages progressive cities to publicize their commitment to biodiversity conservation and management and set targets as part of a strategic plan (ICLEI 2011). However, even the plans flowing from a Durban Commitment often lack specific quantifiable targets. This lack of targets may be attributed to the difficulty of quantifying urban biodiversity in general.

5.3. 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 ecosystem services such as biodiversity, delivered by urban green infrastructure. This intent to produce can be encouraged, and the actual drafting of such a plan can be facilitated by international non-state partnerships. Output takes the form of a strategic plan 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.

Input: Drafting of and intent to commit to policy ideas to maximise ecosystem services provided by urban green infrastructure, commitments (e.g. Durban Commitment), partnership



Output: Cities produce and commit to a strategic plan and or local policies aimed at green infrastructure, conservation of biodiversity and other ecosystem services
(e.g. Local Biodiversity Strategy and Action Plans)

Outcome: Implementation of renaturing initiatives and other biodiversity programmes and projects on the ground

Impact: Actual impact on biodiversity (species and genetic variation), enhancement of other ecosystem services in support of biodiversity

5.4. Past performance: assessing outcome and impact

Similar to the other case studies, the so-called 'funnel-shaped assessment framework' is applied (see *figure 1.5*, Chapter 1, p. 5). Level one refers to simple data, such as the relative number of cities that have published a local biodiversity strategy and action plans, and how many cities are involved with international non-state partnerships or initiatives. Unlike the other case studies level one does not refer to hectares of green space. This is mostly related to the low relative importance of hectares when comparing urban initiatives to state-led initiatives. The second level tries to identify proportional data on those policies that have resulted in actual impact regarding biodiversity conservation. The third level refers to the most specific data, and provides some examples of policies with actual biodiversity impact as identified in level two.

Level 1

The first level assesses the overall number of cities engaged in urban biodiversity policies and renaturing cities agendas. Additionally, it takes into account membership of more general sustainability partnerships. Due to the apparent absence of a comprehensive database on city biodiversity policies, a first stocktaking was performed for megacities and urban agglomerations with a population of over 10 million. Subsequently, this dataset has been expanded to include data on partnerships and to allow for extrapolation. The dataset is built upon the World Urbanization Prospects (WUP) of the United Nations Population Division (UN 2014). These prospects include 1692 cities and urban agglomerations of 300.000 inhabitants or more in 2014. Data resulting from the stocktaking was added to this dataset.

Cities are considered to qualify as having biodiversity as a policy goal when they have produced and published a Local Biodiversity Strategy and Action Plan (LBSAP) or a document of a similar scope. LBSAPs can be considered the local variant of National Biodiversity Strategy and Action Plans as produced under the Convention of Biodiversity (CBD). The document should contain a strategy laying out goals and targets for local biodiversity conservation. Solely plans dedicated to biodiversity are considered. Urban green infrastructure plans are included if biodiversity is the key objective, as opposed to sustainability in general. While this qualification can be considered narrow, it provides a starting point for future comparison between local and state biodiversity policies.

Megacities were selected as the starting point due to their high environmental impact, impact on local biodiversity and high resource consumption. A first inventory was completed of current megacities based on the UN WUP, listing 31 urban agglomerations of over 10 million inhabitants. This inventory was expanded to include the 100 largest cities, ranging in population from just under 4 million to 38 million. Two additional sets were assessed to allow for extrapolation; the 100 largest cities up to 1 million and the 100 largest cities up to 0.5 million. It is expected that, based on the Cities Biodiversity Outlook (CBD 2012), engagement in biodiversity and sustainability policies increases with population size.

In addition to these urban biodiversity policies, a number of other characteristics were taken into account for each city. These characteristics include Durban Commitment signatory, Cities Biodiversity Index participant, and ICLEI membership. The data on these characteristics proved to be sufficient to be applied to the entire dataset. The Durban Commitment: Local Governments for Biodiversity is a commitment to enhance biodiversity at the local level, developed by and for local government. The Durban Commitment entails the pledge to develop and implement a long-term biodiversity strategy. If a city is a signatory to this commitment but has not yet published a LBSAP, it is expected it will do so in the near future. The Cities Biodiversity Index, also known as the Singapore Index, is as self-assessment tool for national governments and local authorities to assist them in benchmarking biodiversity conservation efforts in an urban context. The index comprises 23 indicators and provides a benchmark needed to determine the proportion of urban biodiversity policies with a biodiversity impact in level 2. Finally, ICLEI – Local Governments for Sustainability is a non-profit membership organization which serves as the leading global network of cities, towns, and regions committed to building a sustainable future. Biodiversity is a key agenda of ICLEI, however, the broader focus lies with sustainability. This characteristic gives an indication of cities involved in sustainability as compared to those involved in biodiversity specifically.

Of the 100 largest cities 12 had produced and published an LBSAP or similar document which was available online, 6 cities were indicated to have produced an LBSAP but which was not available online, and 6 cities had signed the Durban Commitment, indicated an LBSAP is forthcoming. Of the 100 largest cities under 1 million 3 had produced and published an LBSAP or similar document which was available online, and 1 city was indicated to have produced an LBSAP but which was not available online. Of the 100 largest cities under 0.5 million 2 had produced and published an LBSAP or similar document which was available online.

Based on these findings it is projected that the percentage of cities with biodiversity as a policy goal is between 12 and 24 per cent for cities larger than 4 million, 3 to 4 per cent for cities close to one million, and around 2 per cent for cities close to 0.5 million have biodiversity as a policy goal.

Table 5.1 *Findings of LSBAPs among the three sets of cities, based on the dataset produced for this case study. *Three of the cities in the initial calculations are below 4 million, but are included here as part of the original set of 100 cities.*

City size	Sample size	LBSAP verified	LBSAP indicated	Durban Commit.	Projected %
4+ mn	100*	12	6	6	12-24%
1 mn	100	3	1	-	3-4%
0.5 mn	100	2	1	-	2-3%

The findings of LSBAPs among the three sets are used for extrapolation into four categories; cities with a population size between 0.3 and 0.5 million, between 0.5 and 1 million, between 1 and 4 million, and of 4 million and more (4+). This extrapolation into four categories aims to give an insight in the expected percentage of cities with an LSBAP for a given population size. Four categories are used in order to provide projections for the complete range of cities in the UN WUP database.

The lower limit of 0.3 million is based on the lower limit of the UN WUP database. The accompanying percentage is assumed to be zero. Although there is reason to believe this percentage above zero, it was not possible to verify within this case study. Hence setting the percentage for the lower limit at zero was considered on the safe side.

The extrapolations for the first three categories are calculated by taking the average of minimum and maximum projected percentages of the lower and upper limit of a category. For example, for the category of 1-4 million the averages are calculated with the values for cities with a population size of 4+ million and 1 million. That leads to the extrapolated minimum of (12%+3%)/2=7.5% and the extrapolated maximum of (24%+4%)/2=14%. The category of 4+ million inhabitants remains the same as the sample findings, as the sample already includes all cities of 4+ million in the dataset.

Based on these numbers, the projected number of cities with an urban biodiversity policy are calculated. The number of cities in a category, as listed in the UN WUP, are multiplied by the relevant percentage.

Table 5.2 *Extrapolation of findings to four categories of cities, based on the dataset produced for this case study.*

City Size	Projected %
4+ mn	12 - 24%
1 mn	3 - 4%
0.5 mn	2 - 3%
0.3 mn	0 %

Category	Extrapolated min. %	Extrapolated max. %
4+ mn	12%	24%
1 - 4 mn	7.5%	14%
0.5 - 1 mn	2.5%	3.5%
0.3 - 0.5 mn	1%	1.5%

Table 5.3 *Calculation of the number of cities with an urban biodiversity policy in 2016, based on the dataset produced for this case study.*

Category	No. of cities in category (2016)	Extrapolated min. and max. %	Cities with an urban biodiversity policy	
			Minimum	**Maximum**
4+ mn	100	12 - 24%	12	24
1 - 4 mn	412	7.5 - 14%	31	58
0.5 - 1 mn	553	2.5 - 3.5%	14	19
0.3 - 0.5 mn	627	1 - 1.5%	6	9
Total cities	1692 (100%)		63 (4%)	110 (7%)

These calculations indicate that between 63 and 110 cities have urban biodiversity as an explicit policy goal, which constitutes around 4 to 7% of cities over 0.3 million world-wide. In comparison, ICLEI membership among the 1692 cities was determined at 264, around 16% of all cities in the dataset.

Repeating this calculation in terms of population covered by an LBSAP creates the opportunity to extrapolate for the global urban population. The global urban population is estimated to be 3700 million (50% of the 7.4 billion world population). The combined population of the cities of over 0.3 million listed in the WUP is 2292 million. Hence, it is assumed that the remaining urban population in cities of under 0.3 million constitutes 1408 million. We further assume these cities do not have an urban biodiversity policy.

Table 5.4 *Calculation of the urban population covered by urban biodiversity policy. *Contains a rounding error.*

Category	No. of cities in category (2016)	Extrapolated min. and max. %	Cities with an urban biodiversity policy	
			Minimum (million*)	**Maximum (million)**
4+ mn	914	12 - 24%	110	219
1 - 4 mn	755	7.5 - 14%	57	106
0.5 - 1 mn	381	2.5 - 3.5%	10	13
0.3- 0.5 mn	242	1 - 1.5%	2	4
0 - 0.3 mn	1408	0%	0	0
Total cities	3700(100%)		178 (5%)	342 (9%)

These calculations indicate that between 178 and 342 million people are living in a city which has urban biodiversity as an explicit policy goal, which constitutes around 5 to 9% of the global urban population.

Level 2

The second level is meant to provide proportional data on those urban biodiversity policies that have resulted in actual impact with regard to biodiversity conservation. However, due to lack of benchmark data on urban biodiversity to which these polices apply, this proved infeasible within the scope of this case study. Instead, this level is used to identify ways forward that could yield proportional data in the future.

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, global assessment would require uniform data. A number of options for data gathering which may yield the relevant data in the future have surfaced over the course of this case study.

Ecological Footprint Analysis evaluates sustainability in general and has been applied specifically on cities. There have been comparative studies, for instance between Shenyang, China and Kawasaki, Japan (Geng et al. 2014). However, the limited availability and absence of a database of such studies complicates further analysis.

Siemens Green Cities Index is a research project conducted by the Economist Intelligence Unit. It measures approximately 30 indicators, about half of which are quantitative. There are several regional indexes available for specific continents. The similar approach to each continent offers potential for comparison on a global scale. A potential limitation is its focus on sustainability in general as opposed to biodiversity.

Urban green space (UGS) a*vailability* is considered an increasingly important aspect of urban planning and research (Kabish et al. 2016). Aside from contributing to the well-being of urban residents, UGS contributes to the conservation of biodiversity. As such it could be used as a biodiversity indicator. In their study Kabish et al. assess the UGS availability of 299 cities in the European Union. However, UGS impact on biodiversity for these cases is unknown. Data on UGS is also available in the Urban Atlas database of the European Environment Agency and municipal land-use databases, which can be used for future analysis. However, it may take time for recently emerged urban biodiversity policies to take effect and to become measurable in terms of UGS availability.

City Biodiversity Index (CBI), also known as the Singapore Index, is a self-assessment tool which encourages cities to monitor and evaluate biodiversity conservation and enhancement more specifically. Currently about 50 cities are in various stages of providing data for this index. It relies on 23 indicators divided into three components (native biodiversity, ecosystem services provided by biodiversity, governance and management of biodiversity) (Kohsaka et al. 2013). Known applicants of the CBI have been added to the inventory. Based on the current available data it remains difficult to define biodiversity impact, but cities with an LBSAP and CBI application are likely a good opportunity for future analysis and may provide detailed examples for level 3. Provided that more cities apply for the CBI, current cities continue to monitor and update their indexes, and if the data is available to the public, the CBI could prove to be the benchmark and evaluation tool that enables analysis to determine proportional data on urban biodiversity policy impact.

Through this explorative study further considerations emerged for future analysis. Initially this study focused mainly on the largest cities. However, the biggest urban growth is expected to be in small and medium-sized cities, not in megacities (CBD 2012). Moreover, the UN World Urbanization Prospects, on which the case study dataset is built, 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 city proper as unit would probably be more accurate, 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. Finally, another aspect that affected results is that English search terms were used in search of LBSAPs or similar documents. It is considered very likely that there are 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 category. It seems likely that these plans are not published in English and therefore did not appear in the search. Overall this could be considered reason for optimism as future analysis with a more elaborate scope may provide substantial additional results.

Level 3

The third level is meant to provide detailed examples of the policies with actual biodiversity impact as identified in level two. This proved to be quite difficult, as level two merely identifies ways forward that could yield proportional data in the future, rather than identifying policies with an actual biodiversity impact. There have been various studies which indicate positive biodiversity impact as a result of urban green space. But so far these impacts have not been directly attributed to urban biodiversity policies that were the subject of this case study.

Alternatively, the stocktaking allows for the identification of cities worth watching. Dar es Salaam, Tanzania, is expected to have a 99% (+5.3 mn) population increase between 2016 and 2030. Similarly, Lagos, Nigeria is expected to have an urban population increase of 77% (+10.6 mn) over the same period. Both local governments are members of ICLEI and signatories to the Durban Commitment, thereby pledging to complete an LBSAP. Additionally, the combination of having applied the CBI and having an LBSAP provides a list of six cities which represent interesting examples. London, Edinburgh, Singapore, Montréal, Paris and Hong Kong may be among the first to deliver measurable biodiversity impact as a result of a dedicated urban biodiversity policy.

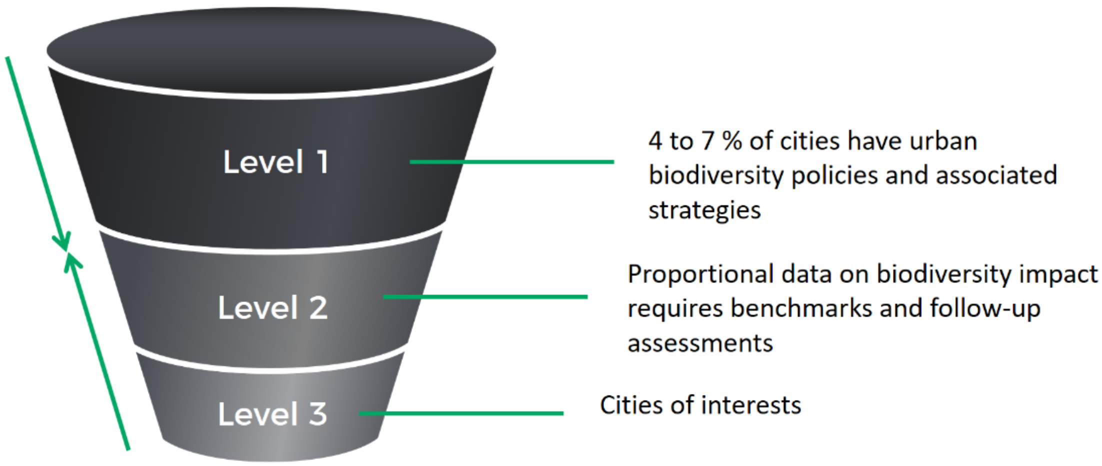


Figure 5.2 *Outcomes and impacts of RNC*

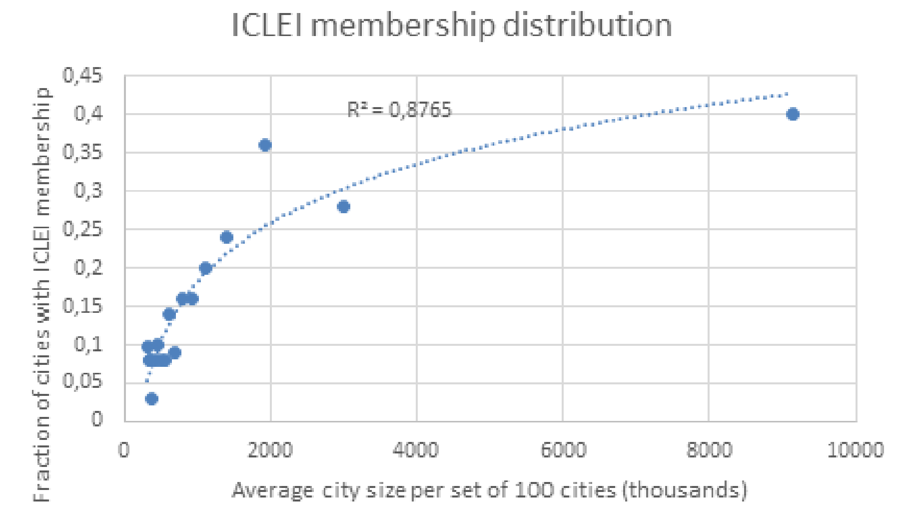
5.5 Expected future performance

In order to project future performance of urban biodiversity policies the calculations from level 1 were repeated while applying three different scenarios for extrapolated percentages and using the UN WUP projected city population for 2030.

The three scenarios vary in terms of extrapolated percentages. The low scenario halves the extrapolated percentages, assuming cities abandon the use of an LBSAP or similar document. The business-as-usual (BAU) scenario uses the current extrapolated percentages. The high scenario doubles the extrapolated percentages and builds on the assumption that relatively more cities engage in urban biodiversity policy through an LBSAP or similar document.

Many of the 1692 cities in the dataset demonstrate population growth towards 2030. This increase leads to a different distribution of cities among categories. The notion that cities changing category has an effect on whether they have LBSAPs is linked to the argument that cities are more likely to be engaged in biodiversity or sustainability policy as they grow in size. This can be supported by looking at the distribution of ICLEI membership and CBI applications among the 1692 cities (Graphs 1 and 2). Both distributions show a reasonable correlation between increase in population size and increase in the fraction of cities with ICLEI membership or CBI application.

Graph 5.1 *ICLEI membership distribution, based on the dataset produced for this case study*



Graph 5.2 *CBI applicant distribution, based on the dataset produced for this case study*

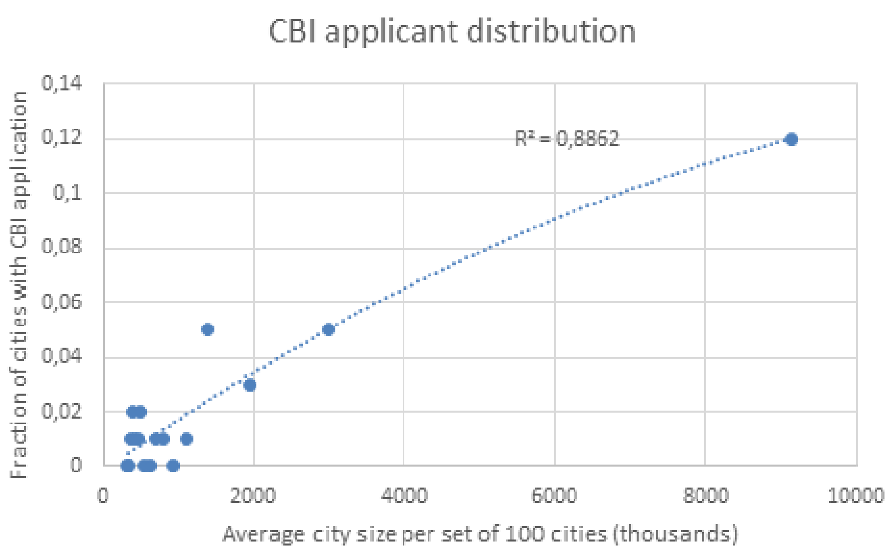


Table 5.5 *Calculation of the number of cities with an urban biodiversity policy in 2030 – Low scenario*

Category	No. of cities in category (2030)	Extrapolated min. and max. %	Cities with an urban biodiversity policy	
			Minimum	**Maximum**
4+ mn	139	6 - 12%	8	17
1 - 4 mn	524	3,75 - 7%	20	37
0.5 - 1 mn	687	1,25 - 1,75%	9	12
0.3- 0.5 mn	342	0,5 - 0,75%	2	3
Total	1692 (100%)		39 (2%)	69 (4%)

Table 5.6 *Calculation of the number of cities with an urban biodiversity policy in 2030 – BAU scenario.*

Category	No. of cities in category (2030)	Extrapolated min. and max. %	Cities with an urban biodiversity policy	
			Minimum	**Maximum**
4+ mn	139	12 - 24%	17	33
1 - 4 mn	524	7.5 - 14%	39	73
0.5 - 1 mn	687	2.5 - 3.5%	17	24
0.3- 0.5 mn	342	1 - 1.5%	3	5
Total	1692 (100%)		77 (5%)	136 (8%)

Table 5.7 *Calculation of the number of cities with an urban biodiversity policy in 2030 – High scenario.*

Category	No. of cities in category (2030)	Extrapolated min. and max. %	Cities with an urban biodiversity policy	
			Minimum	**Maximum**
4+ mn	139	24 - 48%	33	67
1 - 4 mn	524	15 - 28%	79	147
0.5 - 1 mn	687	5 - 7%	34	48
0.3- 0.5 mn	342	2 - 3%	7	10
Total	1692 (100%)		153 (9%)	272 (16%)

The resulting calculations, considering the business-as-usual and high scenarios, suggest that between 77 and 272 cities will have urban biodiversity as a policy goal by 2030, which constitutes around 5 to 16% of cities over 0.3 million. In comparison to the 63 to 110 cities that have urban biodiversity as a policy goal in 2016, this represents an increase ranging from 22 to 147%. The low scenario, while considered unlikely, would present a potential decrease of 38%.

Table 5.8 *Calculation of the urban population covered by urban biodiversity policy. *Contains a rounding error.*

Category	No. of cities in category (2016)	Extrapolated min. and max. %	Cities with an urban biodiversity policy	
			Minimum (million*)	**Maximum (million)**
4+ mn	1322	12 - 24%	159	317
1 - 4 mn	970	7.5 - 14%	73	136
0.5 - 1 mn	483	2.5 - 3.5%	12	17
0.3- 0.5 mn	144	1 - 1.5%	1	2
0 - 0.3 mn	1981	0%	0	0
Total	4900 (100%)		245 (5%)	427 (10%)

If similar calculations are performed for population covered by an LBSAP in 2030, the results suggest that between 245 and 472 million people are living in a city which has urban biodiversity as a policy goal. This constitutes around 5 to 10% of the global urban population in 2030. While the absolute increase relative to the figures of 2016 is nearly 40%, the percentage relative to the global urban population increase is not as significant. This is not surprising, considering the majority of urban population growth is expected in small and medium sized cities.

5.6 Additionality to international biodiversity governance

The 4 to 7% of cities of over 0.3 million which have an urban biodiversity policy are based on LBSAPs, which invites comparison to the National Biodiversity Strategy and Action Plans (NBSAP) as developed and published by the signatories to the Convention on Biodiversity. Currently 193 (98%) of the 196 parties to the CBD have an NBSAP developed or under development (CBD, 2016). Additionality may occur as a 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.

6 Voluntary Sustainability Standards (VSSs), with specific focus on forests

6.1. Introduction to Voluntary Sustainability Standards[1]

Over the last two decades, there has been a rapid increase in Voluntary Sustainability Standards (VSS) for agro-commodities, timber and fish. Starting early 1990s these initiatives were originally initiated by civil society (environmental and social/developmental) NGOs in industrialised countries, often in collaboration with market parties, aiming to raise awareness amongst conscious consumers to buy more sustainable products. They did this by setting standards for improved production, by working with local producers, and by introducing product labels to influence consumer choice. Over time these initiatives were also taken up by front-runner businesses and gradually the type and number of commodities and products for which standards are set and implemented, and labels introduced increased. Sustainability standards and certification have by now become more and more mainstream, with different actors entering the stage and new standards benefiting from (soft) infrastructure being in place. In short: VSS have become a recognised mechanism to connect consumption, production and trade with the aim to create more sustainable development outcomes and impacts (Boström & Klintman 2008; Gereffi et al. 2005; Giovannucci & Ponte 2005; Potts et al. 2014).

As agriculture and deforestation are the largest drivers of biodiversity loss, VSS are a highly relevant development for biodiversity governance. Generally, agricultural expansion is the major cause of land use changes and with increasing intensity of agricultural production (agro-)biodiversity levels decrease. In the last ten years two-thirds of deforestation has been driven by demand for palm oil, soy and beef, as well as timber harvesting (clear-cutting). VSS focus on best practices in production units that include production methods, levels of intensity and location choice to safeguard and improve social and environmental conditions, including better biodiversity outcomes. These best practices aim to reduce agricultural expansion as well to reduce the pressures of agriculture on biodiversity, and to increase (agro)biodiversity levels in the production unit. In some VSS also the relation to High Conservation Value areas is established to improve their protection, and some VSS are designed to also making a contribution to nature conservation in the wider landscape (van Oorschot et al. 2014).

Over the past years VSS have gained a prominent position in approaches to achieve market transformations towards sustainability, but their non-state character implies that their authority is not automatically granted (Cashore 2002). Their added value in fostering sustainability in global value chains is therefore to be constantly assessed and judged (Schouten & Glasbergen 2011) (Gulbrandsen & Auld 2016; Schouten 2013). While most existing sustainability standards address many key biodiversity issues , their contribution

1 This note builds on “Private Meta-governance in Voluntary Sustainability Standards . The case of the ISEAL-alliance”. Marcel T.J. Kok, Mark van Oorschot; furthermore Milder et al. (2015) is a key reference.

to biodiversity improvement is not yet clear. A comprehensive research agenda for standards impact on biodiversity, including different temporal and spatial scales, has recently been put together by (Milder et al., 2015). Although quite extensive reporting is taking place on the state of sustainability markets (including market data for both standards organisations, coverage of criteria in standards as well as market shares of VSS in selected commodities) with also more explicit attention to the contribution of VSS to biodiversity conservation, an absence of performance requirements and impact data makes it challenging for policy-makers to determine where standards are effective in preventing biodiversity loss (Potts et al. 2016).

The analysis in this paper, while starting with a general overview of the outcomes and impacts of VSS in agro-commodities, forestry and fisheries, zooms in on forests, as to allow further linkage with the case-studies on community forestry initiatives, restoration (and zero-deforestation) to bring together the overall synthesis of this report for forests.

6.2. Motives, goals and targets

Milder et al.(2015) provide a succinct summary of motives and goals of VSS from a biodiversity perspective, also recognising the broader goals of reducing social and environmental impacts and providing a market incentive for more sustainable production. These we have expanded to make them also relevant for forestry and fisheries:

Motives

The expansion and intensification of agriculture to meet growing demand for food, feed, fibre and fuel, is a major cause of tropical biodiversity loss and ecosystem degradation. These effects occur either directly, through the conversion of natural ecosystems to cropland or pastures and the degradation of on-farm habitats, or indirectly, through habitat fragmentation, water pollution or diversion, spread of invasive species, greenhouse gas emissions, and other off-farm environmental impacts. Agriculture may also disrupt a range of ecosystem services, from water cycle regulation to soil protection, that underpin food production and other aspects of human well-being (Milder et al., 2015). More sustainable management of forests and fisheries will have positive impacts on livelihoods and the environment.

Goals

Sustainability standards and certification serve to differentiate and provide market recognition to goods (agricultural commodities, timber, fish) produced in accordance with social and environmental good practices, typically including practices to protect biodiversity (Milder et al., 2015).

Sustainability standards furthermore serve to codify the practice of sustainable agriculture, forestry and fisheries in ways that could support the widely held objective of increasing agricultural output while reducing agriculture's ecological impacts and reducing the decline in forest resources and fish stocks and supporting rural livelihood. Such standards also provide a market mechanism to convert demand for more sustainable products into farm/forest/fisheries management-level incentives (Milder et al., 2015).

Targets

VSS typically aim to relate the internationally agreed agreements and norms in their standards (Vermeulen & Kok, 2012), but what is considered as sustainable is defined within the context of specific standards and here different definitions and operationalisations exist resulting in different degrees of stringency and variations in scope (i.e. issues being included).

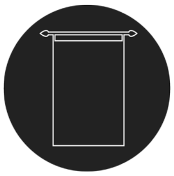
For biodiversity, for biodiversity and agriculture, and for forestry and fisheries the following Aichi targets are relevant: target 5 'Rate of forest loss and rate of habitat loss halved in 2020 and, if feasible, close to zero' and target 7 'Making agriculture, fishery and forestry fully sustainable by 2020'.

Within the Sustainable Development Goals (SDGs), the following goals and targets will become relevant for VSS. SDG 15 on 'Life above land' aims by 2020 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'.

6.3. Theory of change: from input to impact

We apply an input-output-outcome-impact framework to describe the theory of change.

Input: VSS-organisations are non-state, private actors who steer towards global sustainability through global value chains (Cashore, 2002). Their input consists of efforts to develop instruments to reduce social and environmental impacts and provide a market incentive for more sustainable production. These instruments go beyond traditional state-driven mechanisms for international cooperation.



Output: Over recent years, as an output of these efforts, a variety of principles, criteria and indicators are defined and laid down in standards, mechanisms for certification and labelling as well as procedures for (third-party) verification and compliance have been developed in order to guide and assess whether agriculture, forestry and fishery is happening in a sustainable manner. These mechanisms underlying VSSs have been institutionalised in several organisations for either standard formulation and revision; independent (third-party) auditing; accreditation of auditors; assurance systems for supply-chain tracking, etcetera. Together, these outcomes can be characterised as the "soft" infrastructure for implementing voluntary sustainable production practices.

Outcome: The outcomes of VSS have to become visible in the application of more sustainable practices by farmers, foresters, fisherman. Indicators showing these behavioural changes are for instance the area under certified sustainable production of each commodity, the number of farmers educated on better practices, the share of production of a specific commodity being covered by certification schemes, and the share of consumption covered by consumer or business-to-business labels (ie market uptake).

Impacts: for biodiversity of such altered management practices distinguish between on- and off site impacts and between managed (agricultural systems) and (semi)natural systems (forests, oceans); intermediate results that distinguish between conservation of natural areas, conservation value of managed systems being improved and pollution and off-site impacts being reduced; and broader, less direct impacts over larger spatial and temporal scales, including avoided conversion of natural ecosystem . Furthermore indirect impacts of VSSs are interesting to take into account, for example responses by governments to step in with regulations or developing domestic standards inspired by an internationally accepted VSS (like the Indonesian Palm Oil Standard in response to RSPO).

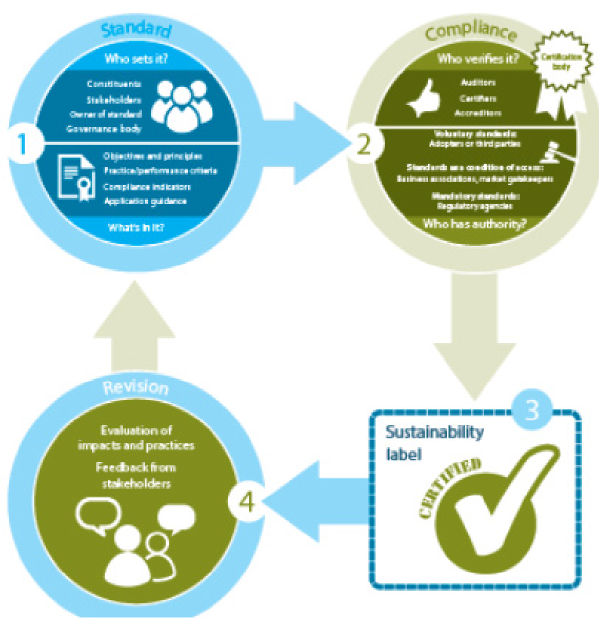


Figure 6.1 *The working of certification systems (adapted from (SCSKASC 2012)*

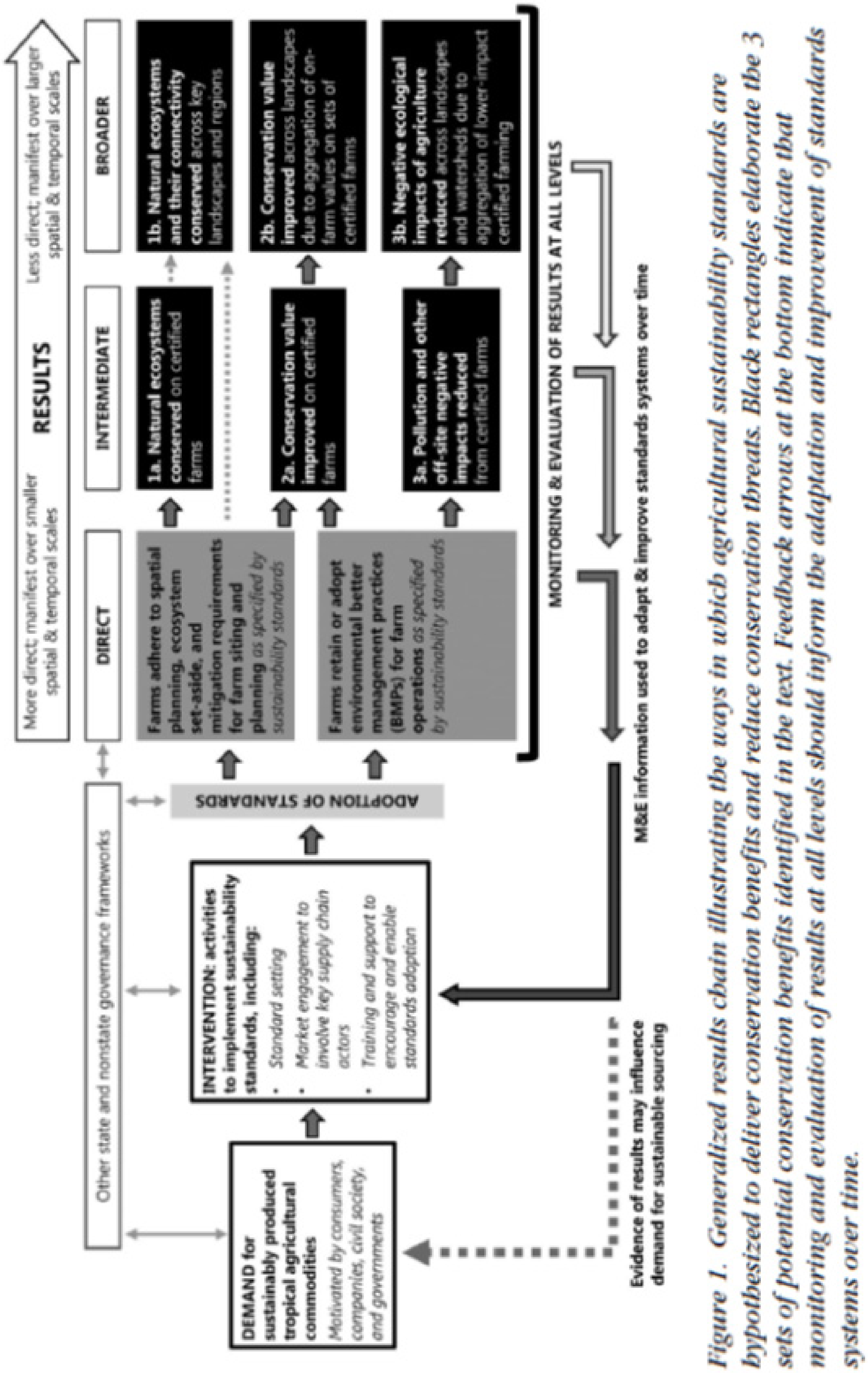


Figure 6.2 *A theory of change for agricultural sustainability standards to deliver conservation benefits*

Figure 6.2 above provides an theory of change for agricultural sustainability standards to deliver conservation benefits at different levels (from Milder et al., 2015).

We analyse past performance following the I-O-O-I framework giving an 'overall' assessment of the contribution of VSS to the conservation of biodiversity, that we further elaborate for certification in forestry in the next section using the 'funnel-model' inspired by Milder et al. (2015).

Output - "soft infrastructure in place"

Parties involved in specific supply chains, often through multi-stakeholder, round tables, have agreed upon production and management standards that comprise sets of criteria for the sustainable production, processing and trade of commodities. Certification plays an important role in the implementation of production standards, providing a means of verification as well as credibility to sustainability claims for the market. Certification distinguishes between the certification of the production process and management on the one hand, and chain of custody certification for the trade in sustainable agricultural products, timber and fish (which traces their origins), on the other. Each business actively participating in a supply chain must be audited by a third party auditing agency. If they cannot meet the requirements of a standard, they will first have to improve their operation and production processes. The auditors themselves have to be accredited by the organisations that have developed the standards. The ISEAL Alliance (Alliance for International Social and Environmental Accreditation and Labelling) for example, has developed good practice codes for the appropriate development, control and evaluation of standards (ISEAL 2014). There are also private standards audited by businesses themselves or they audit each other, and there are also examples were certification is carried out by local stakeholders.

In summary, two decades of development and application of VSS has created an extensive, 'soft' infrastructure of standards and certification, used by a large variety of actors and initiatives in the market, allowing new standards to be more easily implemented in specific production areas or existing standards to be expanded to other commodities. Certification has thus taken on a dominant role within the current strategies that make supply chains sustainable. Businesses and governments are creating demand by imposing certain requirements on suppliers (exporters, processors and producers) in their purchasing policies – sometimes even referring to specific standards and their certification labels (such as Fair Trade, UTZ Certified, MSC, FSC).

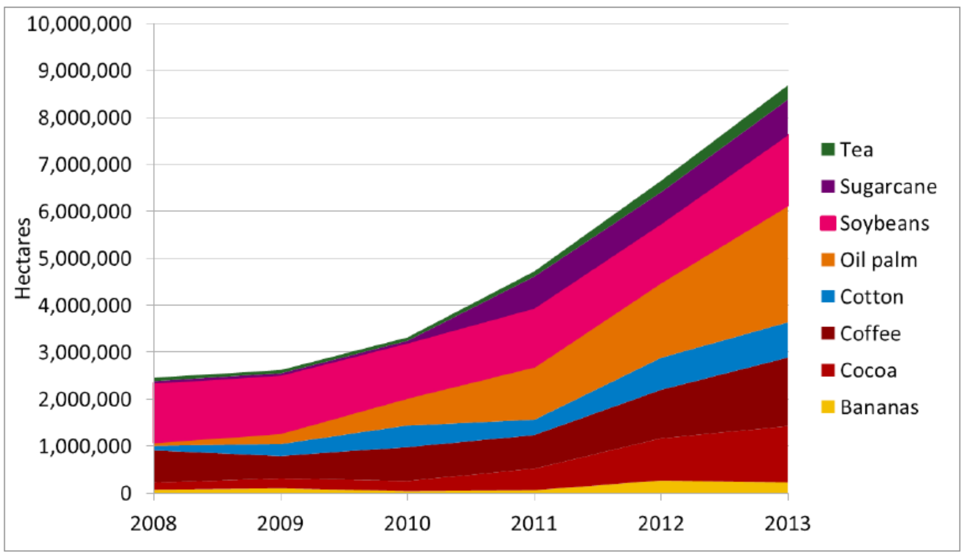
Outcome – VSS becoming mainstream

In itself an illustration of infrastructure being in place, the VSS themselves started to report on the outcomes they create. This is happening through the State of the Sustainability Reviews (Lernoud et al. 2015; Potts et al. 2014; Potts et al. 2016) that provide an overview of trends in the market. Key findings on outcomes include:

- Sustainability standards continue to experience rapid expansion and have penetrated mainstream markets and the reviews document a persistent trend towards sustainable sourcing (see *figure 6.3* for agro-commodities).
- Agricultural production compliant with these standards has grown at an average of 35 per cent per annum between 2008 and 2014.
- Standards already cover a significant portion of some commodity markets. Half of global coffee production, 25 to 30 percent of wood production; 30 per cent of cocoa production, 22 per cent of palm oil production and 18 per cent of global tea production is standard compliant; however, standard compliant production only accounts for a small portion of total global agricultural land area with minimal presence in major staple crops.
- Sustainable markets continue to be defined by persistent over-supply of standard compliant production (so buyers have ample choice – which is a positive outcome; but it may result in downward price pressure – that would be a negative outcome).

- Production for sustainable markets is concentrated in more advanced, export oriented economies (so not necessarily in areas where it is most needed, either to contribute to poverty reduction or to protect biodiversity).
- Average criteria coverage of VSS is declining as standards target mainstream markets; newer mainstream-oriented standards apply criteria of reduced depth and breadth as a means for allowing for more rapid uptake and hence limiting potential impacts
- The distribution of compliant production is primarily determined by where compliance costs are lowest rather than where need is highest.

Figure 1: Development of the VSS compliant area worldwide, 2008-2013 (eight selected commodities, minimum possible)



Sources: FiBL-IISD-ITC survey, 2015: 4C Association, 2014 and 2015; Better Cotton Initiative (BCI), 2014 and 2015; Bonsucro, 2014 and 2015; Cotton Made in Africa (CmiA), 2014 and 2015; Fairtrade International, 2014 and 2015; GLOBALG.A.P., 2015; FIBL, 2015; ProTerra Foundation, 2014 and 2015; Rainforest Alliance/SAN, 2014 and 2015; Roundtable of Sustainable Palm Oil (RSPO), 2014 and 2015; Round Table for Responsible Soy (RTRS), 2014 and 2015; UTZ Certified, 2014 and 2015.
Note: The data in this graph were not adjusted for multiple certifications. The graph assumes that there is maximum amount of multiple certification occurring within each commodity corresponding to the minimum amount of VSS compliant area per commodity. Therefore, the total amount of VSS compliant area corresponds to the VSS with the largest compliant area operating within a given commodity sector.

Figure 6.3 *Trend in VSS compliant areas in 8 agro-commodities (underestimation, as a maximum amount of multiple certification is assumed)*

Impacts – glass half full or half empty?

While most existing sustainability standards address many key biodiversity issues, an absence of performance requirements for certification to achieve specific biodiversity results and impact monitoring makes it challenging for policy-makers to determine where standards are most effective in preventing or even reverting biodiversity loss (Potts et al., 2016). One of the first reviews on impacts of certification on both environmental and socio-economic conditions is the one by Blackman & River (2010). In summary, it showed mixed results. Some evidence of positive local social effects was found, for instance on farm income and working conditions in forestry. But such positive effects did not generally occur, and also negative effects were reported. For biodiversity, only a small number of well-designed impact studies were available at the time, and most of them treated coffee production (mostly Fair Trade) and certification of forest management (FSC only). A more in-depth analysis of biodiversity effects of certified forest management (van Kuijk et al. 2009) found that, in spite of the large variety in responses between species, the different forest management practices associated with forest certification appear to benefit biodiversity in managed forests. This was later confirmed by expert interviews (Zagt et al. 2010). There was especially consensus on the important role of certification for tropical forests, as it has done more than any other initiative to protect forest habitats and promote sustainable management practices. More on these impacts is given in the section on forest certification.

In response to the generally observed lack of well-designed impact studies, both in terms of experimental design and focus on biodiversity, more and more impact studies have been conducted since. The broad RESOLVE review (SCSKASC 2012) concluded that there is reasonable evidence that certification has had positive impacts on the environment and biodiversity in particular cases. However, the local variability in environmental conditions between sites will affect specific results, and together with methodological limitations, this does not allow simple extrapolation of the findings beyond the immediate cases under review. Some of the differences in types of impacts between sectors may arise from the nature of the resource production processes. In forestry and fisheries, the resource harvesting process is closely tied to the semi-natural status of ecosystems, whereas in cultured or human-designed production systems such as agriculture and aquaculture, the focus of sustainability standards is much more on pollution and waste issues. Impact study reviews should take the specific ecosystem setting clearly into account.

In impact studies on agricultural standards for certification in coffee and cacao, much evidence was found suggesting that the inclusion of environmental criteria may indeed support biodiversity conservation, but it is less clear to what extent certification leads to improved conservation impacts (Tscharntke et al. 2015). Current certification models are mostly oriented towards on-farm improvements, and this will limit the delivery of broader conservation benefits, as maintaining biodiversity depends on processes at higher spatial scales. To address this scale mismatch, initiatives for certifying production at landscape levels are now advocated, for instance by ISEAL (Mallet et al. 2016).

The Sustainable Agriculture Network (SAN) standard is of special interest here, as biodiversity conservation has always been an important focus of the SAN's and Rainforest Alliance's work (Milder et al., 2015). In addition to promoting conservation through the SAN Standard, the SAN members and their local partners provide training on biodiversity-related topics for farmers. The SAN Standard contains criteria for continuous improvement focused on ecosystem conservation and wildlife protection, and several other criteria that directly or indirectly address biodiversity conservation at different levels (both on-farm and landscape wide).

A recent review of impact studies (Milder et al., 2015) concludes that SAN certification contributes to healthier natural ecosystems, both on the farm and in the surrounding landscape. Multiple studies have documented increases in tree cover and wildlife protection on certified farms, relative to non-certified farms or relative to pre-certification conditions. Furthermore, the shade trees, natural ecosystem patches and riparian corridors on certified farms can contribute to conservation in the broader landscape, as found by independent studies in Brazil, Colombia and Ethiopia. Impacts of standards on ecosystem services of natural and cultivated ecosystems have until now not yet been addressed explicitly in standards, although they have the potential to deliver societal benefits through conservation and sustainable use (van Oorschot et al. 2016).

To deal with the lack of impact-research ISEAL has set up a cross-standard platform for discussing and promoting impact research. This can be seen as an attempt of standards organisations through ISEAL to enhance standards credibility by actively approaching and helping members to implement the ISEAL impact measurement code. Some members are active on this agenda themselves, and invest in designing new impact measurement frameworks and methods (for FSC see (Romero et al. 2013); for SAN-RA see Milder et al., 2015; for MSC see (Agnew et al. 2013).

6.4. Past performance: assessing outcome and impact

We now apply the 'funnel' to further analyse the impacts of forest certification. The two major VSS in forests are the Forest Stewardship Council (FSC, established in 1993) and the Programme for the Endorsement of Forest Certification (PEFC, established in 1999). Van Kuijk et al. (2009) mention that assessing the impacts of certified forest management is a complex topic. Forests differ from place to place, and so management practices vary. Differences in logging intensity, logging pattern and timing, the size and variety of species harvested, extraction method and post-harvest treatments all contribute to different responses by plants and animals. So it is necessary to look more closely at what types of forest have been certified up to now. Furthermore, there are a large historical difference in forest laws that determine the changes that voluntary forest certification is able to induce.

Level 1 System wide forest certification monitoring

The analysis in this section is largely based on the State of Sustainability Initiatives 2014 and 2015 (Potts et al., 2014; Lernoud et al., 2015). It is (conservatively) estimated that the two major VSS initiatives had together certified almost 10%, or 387 million hectares, of the worlds forested area by 2014 (using 4 billion hectares in 2011 as number for total global forested area). The purpose of SFM is to apply improved silvicultural practices in forests that are managed for human use and exploitation (like harvesting of wood), so to show the potential influence of VSS it is better to only take production forests into account. The (FAO 2010) estimated that about 1,6 billion hectares of forests are managed for production and multi-functional use, which means that 24% is managed sustainably under the FSC and PEFC standards. And the share continues to grow. Including development up to april 2016, (UNECE & FAO 2016) writes that certified forest covers about 30% of the productive forests, estimating that currently almost 30% of all industrial round wood originates from certified forests. (All these numbers are corrected for double certification: - 15% in the total volume of both systems; see Lernoud et al.; unlike numbers in Potts et al., 2014)[2].

In 2013 71% of certified forest were found in 5 countries in the Northern hemisphere: Canada (40%), US (12%), Russia (9%), Finland 5% and Sweden (5%). Most of the certified managed forest area was in North America (49%) followed by Europe (40%). These figures obviously have raised questions about the aims being achieved in view of social and environmental/biodiversity benefits of forest certification in developing regions. Only 12% of certification is taking place in Africa, South America and Asia. Average certification growth rates between 2009-2013 are 6%, but are higher for tropical regions. See trend *figures 6.6 and 6.7* below.

Although the major part of all deforestation and illegal logging has occurred in tropical forests over the last three decades, only 3 countries in the top 15 of certified countries contain tropical forests (Australia, Brasil and Malaysia). So forest certification is disproportionally concentrated in Northern developed economies (88% of certified forests and only 34% of the world's forest) and largely not happening in the countries that see a decline in forest area (see *figure 6.4* on the next page).

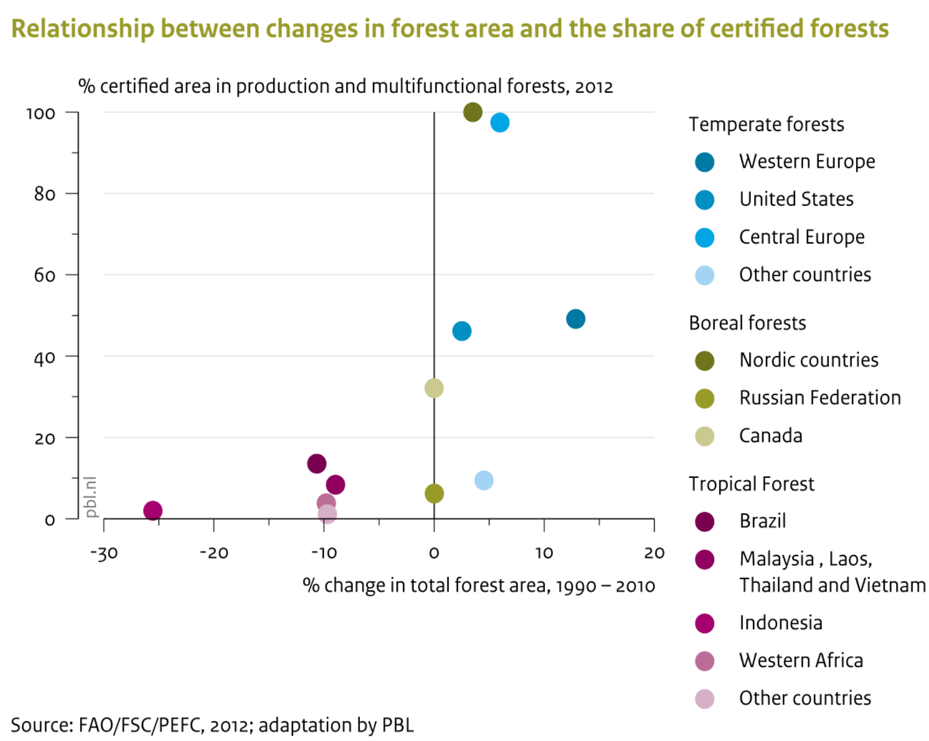


Figure 6.4 *Presence of forest certification in 2013 compared with the net change in forested area from 1992-2011*

2 The percentages would be slightly higher if we would also use Global Forest Cover (RRI, 2014) here as well.

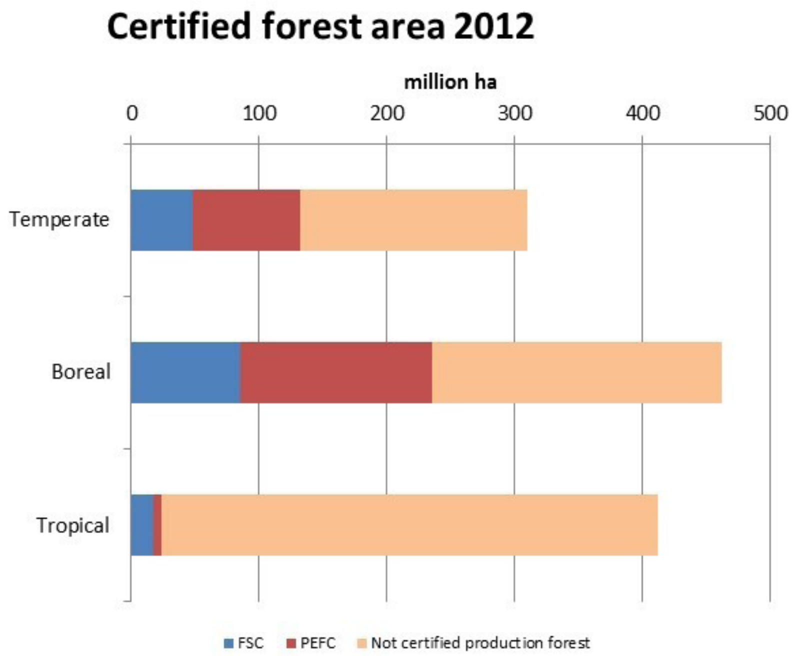


Figure 6.5 *Area of FSC and PEFC certified forests, and uncertified area in for forests managed for production purposes in different climatic zones.*

Figures on FSC certification per climatic region, biome and forest type are also available and reflect the challenges to tropical certification, but also show that the FSC system is mainly implemented in natural and semi-natural/mixed forests (*Figure 6.5*). Plantations are much less present in the FSC coverage. For PEFC, data per climatic region is also available. The 2012 data are showing a focus on temperate and boreal regions, with hardly any certified area in the tropics: 61% of PEFC certified area is in the temperate zone, 36% in the boreal zone and only 2% in the tropics (data from PEFC international; in van Oorschot et al 2015). Data on PEFC coverage per forest type is not available, but given their historical focus on conventional forestry systems in Western countries, most of it will probably be on semi-natural forest with rotational felling, where regrowth occurs more or less natural, and plantations where regrowth is done by planting (often with species exotic to the climatic region) and intensive management practices.

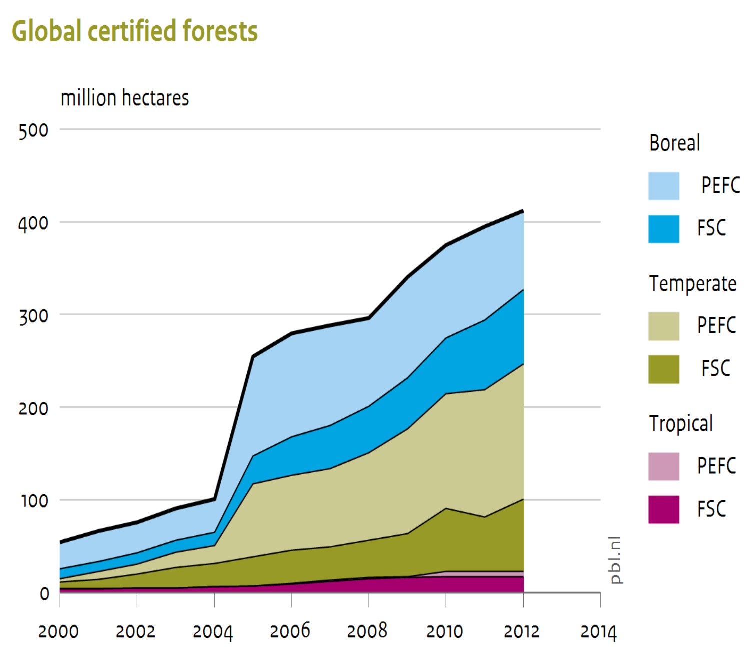
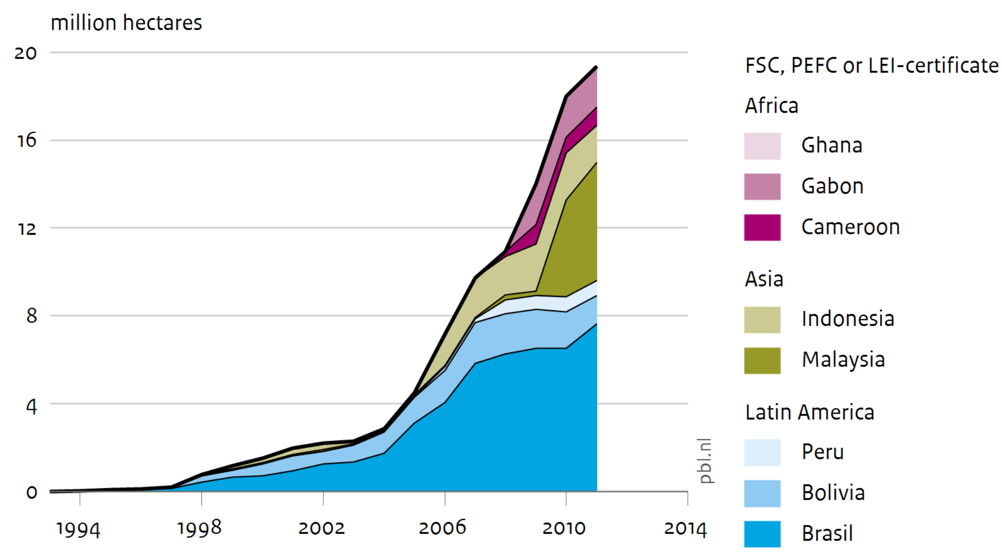


Figure 6.6 *Trends in forest area certified for sustainable management (SFM). Growth in certified area has been most prominent in temperate and boreal regions. (Source: FSC data from FSC-database and (PWC & IDH 2012); PEFC data requested for van (van Oorschot et al. 2015).*

Global certified tropial forests



Source: FSC; PEFC

Figure 6.7 *Certified area in tropical regions shows a higher than average growth rate since 2004, but is still low in absolute numbers. (Source: FSC data from FSC-database and PWC/IDH 2012; PEFC data requested for van Oorschot et al., 2015).*

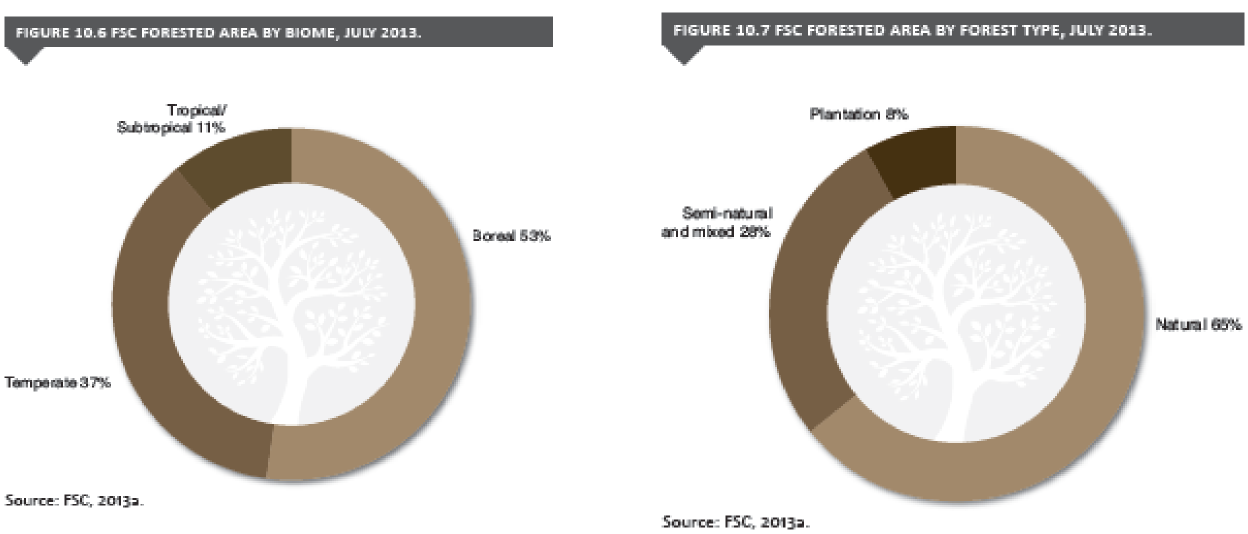


Figure 6.8 *FSC forested area by biome and forest type, july 2013 (Potts et al., 2014).*

Table 6.1 *Estimates on global certified forest area, distributed over different climate regions and forest management types. Overall, the certified area is dominated by semi-natural boreal and temperate forests where rotational felling is practiced. Plantation forestry and selective logging are much less represented in the global certified area.*

FOREST AREAS		***Tropics***		***Temperate***		***Boreal***			***World***
million ha		SFM type	area	SFM type	area	SFM type	area		Certified estimate
Natural forests	**Selective logging**	80%	18	20%	39	0%	-		57
Semi-natural	**Rotational felling**	0%	-	50%	97	100%	172		268
Plantations	**Rotational felling**	20%	5	30%	58	0%	-		63
			23		193		172		388
Certified area - regional distribution									
FSC area			11%		32%		57%		150
			17		48		86		
PEFC area			2%		61%		36%		237
			6		146		86		
total			6%		50%		45%		for year 2012
Sources:									
van Oorschot et al 2015 - Handelsketens onder de loep									
Assumptions on SFM types - derived from inventory of wood production systems (Eric Arets et al, 2010)									
inventory data obtained from FSC and PEFC (see van Oorschot et al 2015)									

With this data on coverage of climatic regions, and the distribution over different management systems, a first estimate of areas under each form of SFM can be constructed (see *Table 6.1.* above). This is a necessary step as certification effects depend on forest and management types.

Level 2 Management changes induced by forest certification

An important aspect to consider in determining the impacts (= stimulated change) of certification is that improvements that have to implemented to comply to the certification criteria can be costly. The costs for implementing sustainable forest management (SFM) are a well-known obstacle (PWC and IDH, 2012). Especially in the tropics the costs are high, as more changes are required here to comply compared to regions with a long history of formal forest management laws. The financial results of selective logging may even decline when sustainability criteria prescribe lower annual allowed cuts, making this less attractive business (Arets & Veneklaas , 2014). The high costs of implementing changes in tropical regions also implies that more impact can be expected.

Costs being a barrier for certification in a sector where price premiums are low or even absent has several consequences for creating impact. Certification has first taken off where management was close to criteria and certification costs were relatively low, but the induced and achieved changes have logically been modest. The effect is also referred to as reaping low-hanging fruit (Cashore & Auld 2012; Gullison 2003); see also Potts et al., 2016 for a similar argument on agro-commodities). It is attributed to the presence of good forest management practices laid down in national forest laws (Cashore et al. 2003). As a consequence, certification has grown relatively fast in temperate and boreal countries (with Russia making fast progress in more recent times), while tropical regions are still behind (see also *figure 6.5*).

Still, a broad overview on FSC certification (Karmann & Smith 2009) mentions a large number of verified positive impacts on forest management and changes in governance structure. According to these authors, certification has clearly contributed to practical improvements and is not just a confirmation of already existing management practices. For instance, the identification of HCVAs in certified concession can be labelled as an impact, as they are additional to the areas under formal national biodiversity protection policies (Elbakidze et al. 2011). There is also other evidence from comparative and experimental case-studies that show positive biodiversity changes in response to certified management (see level 3 cases).

There is also evidence of management improvements made during the certification, based on so-called CARs research - Corrective Actions Request (Peña-Claros et al. 2009). This type of research analyses compliance failures in certification processes, and found that recorded corrections were relatively even distributed over the different domains of sustainability, including indicators relevant for biodiversity. However, this type of research can not show the ultimate effects on environmental conditions and biodiversity as a response to the correction requests. At present, certification processes based on performance indicators (final impacts) are promoted, instead process indicators (outcome results).

With upscaling of forest certification, reaping the higher hanging fruits can be expected. More change and impact can be induced but this depends on the availability of additional funds, to cover the higher costs. Another possibility is to raise the bar for all producers (PWC &IDH, 2012). The existence of good national forest governance is seen as an important aspect of enabling conditions for successful certification (Cashore, et al. 2003). Special EU supporting programs in the form of bi-lateral 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 also create a level-playing field for all producers on legal aspects, and not just for the ones seeking certification. These VPA programs (voluntary partnership agreements) are now being implemented for countries that are relevant for EU imports: Cameroon, Central African Republic, Ghana, Indonesia, Liberia and the Republic of the Congo.

The absence of sector wide and representative impact and performance monitoring makes it at present impossible to produce a reliable estimate of the share of certified area where improved forest management has induced positive biodiversity impacts.

A first rough estimate could be produced making some crude assumptions, such as: assuming no effect of low-hanging fruit, and therefore excluding the area of certified forest in countries with long standing national forest laws; assuming effective forest protection through certification, and therefore labelling the area identified as voluntary HCVA identification as additional protection effect; assuming positive changes in forest management in response to CARs, and estimate the long-term effects of management improvements. To produce such an estimate with a reasonable degree of certainty will require further research, and this is not done within the context of this research paper.

Level 3 Focused impact research – estimating the technical potential

Impacts on biodiversity of certifying forest management

Now the forest area where sustainable management is practiced is known (outcome at level 1), the step we take here is to describe and estimate the potential impacts of certification on forest biodiversity. We speak of “technical potential” here, and will estimate the differences in biodiversity between conventional practices and responsible practices prescribed in standards for SFM.

Certification according to the VSS criteria can have positive impacts on biodiversity conservation or even restoration of biodiversity in the forest management unit. The principles of both the FSC and PEFC standards contain several relevant aspects. In the FSC standard (V5-2; FSC 2015), principle 6 describes the necessary actions that must be taken to monitor, maintain and conserve, and even restore environmental values, including rare and threatened species and habitats. Principle 9 describes that a precautionary approach should be taken for identifying and maintaining those parts of the forest management unit that qualify as High Conservation Value Areas. Forest plantations, that are usually low in biodiversity values, can only be certified when they are not established on areas converted from natural forests (threshold date: after 1995). The PEFC principles and criteria contain similar requirements, for instance: Criterion 5.1.11 Conversion of forests ……, 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…; Criterion 5.4.13 Standing and fallen dead wood, hollow trees, old groves and special rare tree species shall be left…. (see for complete text: PEFC ST 1003:2010).

Different types of potential impacts and change mechanisms can thus be distinguished. To start with, the biodiversity value of production forests largely depends on the intensity of forest-use and management (Alkemade et al. 2009; Burivalova et al. 2014). High forest biodiversity values are commonly found in natural and secondary forests with a protection status and a low use-intensity. Natural tropical forests are especially important for biodiversity conservation, as high biodiversity values are found in tropical moist forests (Gibson et al. 2011). In secondary or semi-natural forests that are managed for multiple purposes, relatively good biodiversity values can be found, depending on the management intensity and vice versa on their naturalness. Some argue that these forests are important for forest biodiversity as they are extensive, contain a large part of species that can also be found in natural forests, and at the same time their commercial use provides a financial basis for their existence and maintenance (Dent & Joseph Wright 2009). On the other extreme, not a lot of biodiversity values are commonly found in plantations, because they are planted with exotic species and managed intensively for the sole purpose of wood production. A disproportionately high share of wood is produced in plantations nowadays (Carle & Holmgren 2008). This high productivity means that less area is required to fulfil the economic demand for wood, and so they can play a role in conserving natural forests in other locations (so-called sparing hypothesis). Their potential indirect effect on biodiversity can only be assessed in a broader context, both spatial and in terms of governance.

Following the general scheme of (Milder et al., 2015) (*figure 1.5*), on-site, off-site and broader biodiversity impacts of certifying forest management may occur, and these have different levels of probability and certainty. There are several main effects to consider and estimate:

1. Most **natural and recovered mixed forests** in the tropics can be used for selective logging. This means that wood is produced by logging specific individual trees of a commercially valuable species. A characteristic feature is that wood production per hectare is low, and that the forest is degraded through irresponsible logging practices. 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). In applying so-called RIL practices (reduced impact logging) several techniques are practiced to avoid collateral damage of logging and 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 reduces damage on commercially not important 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 practices, and it is emphasized that this should be performed by appropriately trained crews (Putz et al. 2008). This is a clear example of direct effects of on-site management improvements.

There are hardly any experimental studies available that compare conventional logging techniques with certified forest management. But there is a lot of research available on specific silvicultural techniques that are contained and prescribed in certification standards (van Kuijk et al 2009). So for the sake of simplicity, SFM in selective logged forests is equated here by implementing RIL and other supporting techniques (Peña-Claros et al. 2008). A meta-analysis based on >100 publications revealed, next to substantial variability, that if collateral damage is reduced and more sustainable silvicultural treatments are applied that: timber yields after the first harvest cycle can be sustained, although at a lower level; three quarter of carbon can be retained in once-logged forests; 85 to 100% of species of mammals, birds, invertebrates, and plants remain after logging (Putz et al. 2012). Although there is still a lot of discussion about the effectiveness and efficiency of RIL techniques, we conclude here that there is enough evidence available for attributing positive on-site biodiversity effects to implementing SFM in selectively logged forests.

Effect estimate

Here, we will produce a first estimate for this positive effect, by using the Mean Species Abundance (MSA) biodiversity indicator developed by PBL for use in their global biodiversity model GLOBIO (www.globio.info; Alkemade et al., 2009; for further explanation see text in *Annex 4* on p.108). The indicator gives a local indication of the 'naturalness', i.e. the degree to which local biodiversity resembles a natural un-impacted situation. This is a useful indicator for this exercise, as it can easily be combined with area data. Impact indices have been derived for several land-use types, based on literature comparing un-impacted an impacted forests. The MSA indices can be multiplied with the area of a certain biome for which the impact is representative (giving a quantity with an area dimension, and a unit commonly expressed as hectare. MSA). This results in impact weighted summed areas, which is useful for comparing different options in scenario analysis (see for instance (ten Brink et al. 2010).

In (Schippers et al. 2016), the residual biodiversity in lightly used forests is set at 70% MSA. This means that about 70% of the species populations that are usually found in unaltered natural forests can still be found. A literature review by (Arets & Veeneklaas 2014) contains several estimates on RIL effects. In South American tropical forests, logging damage is on average 17% for RIL and 32% for conventional selective logging (CL) while in South East Asia, with higher logging intensities, this is on average 54% for CL and 28% for RIL. Based on this review, a modest positive effect of plus 10 to 20 %-point higher biodiversity is attributed to RIL compared to CL, roughly halving the biodiversity loss.

2. In **semi-natural secondary forests** that are widespread in the temperate regions (Europe, North-America, China, etc) , the most common conventional production system is rotation forestry, where large areas of forest are cleared and regrowth is assisted by silvicultural measures. Depending on the replanting schedules (native species) and other measures, these forests still support a relevant amount of forest species. This type of forestry can be distinguished from artificial plantations by the used species (natives versus exotics) 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, corridors connecting forest patches, riparian buffer zones, HCVA identification and set-aside requirements. This general positive outcome is confirmed by interviews with scientific experts, forest certifiers and forest managers (Zagt et al 2010). (Karmann & Smith 2009) also mention that there are improvements attained in temperate forests as a result of FSC certification. This is also an example of direct effects of on-site management improvements.

No quantified data could be derived from these qualitative literature reviews, so rough assumptions are needed for a first-order estimate. In Schippers et al (2016), the residual biodiversity in these intensively used forests is estimated at 50% MSA. The evidence base is clear enough to award the certified practices with a relative modest positive potential biodiversity effect of 10 to 20%-point (very uncertain), reducing the impact on average by about a third.

3. An essential criterion for certifying **forest plantations** is the requirement that a plantation may not be established by **converting natural forests**. The on-site effect of establishing a plantation will then depend on the preceding land use. When plantations are established on abandoned agricultural lands, MSA biodiversity values will probably rise, although this will depend on the type of trees used (native or exotic species; in mono-specific stands or mixed plantations). This is an example of forest restoration. Using the general biodiversity values of the different GLOBIO land-use classes (Schippers et al 2016; expressed in the MSA biodiversity indicator), a change from intensively used cropland (10% MSA) to a forest plantation (30% MSA) would result in an absolute biodiversity increase of 20 %points. But changing from selective logged natural forests (which is by definition not a natural forest) to partial plantation forestry would mean an decrease of 40 %-points. There is no information available on the type and frequency of land-use changes related to plantation establishment and certification, apart from local examples (see for instance (Reynolds et al. 2011). The worldwide occurrence of these land-use change mechanisms are, to our knowledge, unknown.

We assume for the sake of simplicity here that half of the certified plantations are established on degraded agricultural lands. And the other half is assumed to be established on selective logged forest, where HCVA identification must take place (see further under option 4).

4. During the process to become certified, it is obliged to undertake the **identification** and protection of areas with special conservation values (HCVA). In the definition of what is valuable, biodiversity is a prominent aspect next to other forest values like community needs. Locally important ecosystem services

are also recognized. There is no fixed concessions area share for this criterion. From a limited number of cases studies on forest plantations, a wide range of set-aside percentages can be derived. The cases showed that 5-40% of the total FMU area generally falls under the HCVA definition (NGPP & WWF 2009), and this depends on the forest structure and history, local conservation priorities, and national value definitions. Identification as a HCVA does not automatically mean that forests will be fully protected, low intensity use is still possible. The former land-use is not purely undisturbed forest, as that is not allowed under the FSC and PEFC criteria for plantation establishment. This "avoided loss" effect is in the end quite modest, as it only applies to a small part of the certified forest management area.

5. Another effect of forest plantations is the possible land sparing effect. Plantations have a high productivity, and this intensified wood production concentrates the production function on a limited area, making way for larger areas of forest to remain semi-natural forest('avoided loss') that can recover to a more natural state ('regrowth'). Calculations showed 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 & Veneklaas, 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). Again, there is no information available on land-use changes and possible sparing effects related to plantation establishment on a regional scale.
A hypothetical calculation of the sparing effect can be made, assuming that certified plantations are established on an area with abandoned extensively used crop land, and that the spared land contained selectively (extensively) logged forest that can now restore to a more natural status. For the regional forest system as a whole, this way of plantation establishment results in a positive MSA effect of over 30% for Southeast Asia and almost 40% percent in South America.

This sparing effect cannot simply be attributed to plantation managers holding a certificate for SFM, as they do not control the indirect wider regional effects. Controlling regional land-use and allocation of plantation establishment is under the control of local and national government bodies. However, through trade, consuming countries may exert an indirect influence on land-use planning in producing countries, for instance by partnerships to improve the status of forest laws and enforcement with the aim to bring these countries up to an acceptable level of forest governance. Such policies have emerged for instance in the EU under the influence of NGOs and voluntary production standards, so there is a potential indirect effect through government interaction. This can be further promoted by public procurement criteria in consuming countries, in which the use of VSSs are a prominent element.

So the calculated potentials of the sparing effect have a very low certainty, much depends on complementary forest governance. Therefore, they will not be taken up here in a quantitative way, as there are no clear indications of a direct certification effect.

In the above discussed and quantified effects, comparisons are used between conventionally managed and sustainably managed forests. But this gives a somewhat artificial and optimistic idea of forest change brought about by the certification process. To obtain a reliable impression of additional certification effects, data on pre-certification forest biodiversity status and monitored effects of improved management is needed. However, there exist no general monitoring system that tracks and reports the initial status and the improvements realized at this moment.

When we apply all these estimates on the different certification effects to the distinguished certified forest areas from ***Table 1***, the total potential biodiversity effect of forest certification can be calculated by multiplying the estimated effects with the appropriate certified forest area. In this way, we can compare the positive biodiversity effects of certified sustainable forest management with the biodiversity loss of conventional forestry (*Table 6.2.*). A full uncertainty analysis is at this point unfeasible, as there is no information on probability distributions. But it is unlikely that all effect estimates lie either at minimum or maximum values, so it is acceptable to look at mean effects.

The estimate of impacts of conventional forestry in the now certified are serves as a reference here (top half of the Table). The impacts are calculated for the direct land-use effects of forestry, while other more indirect effects (like fragmentation and habitation at forest fringes are not taken in to account (like what is done in GLOBIO scenario analyses, Kok et al. 2014). This results in a total loss of 195 million ha·MSA in

Table 6.2 *Comparison of biodiversity losses due to conventional forest exploitation (top) and potential positive biodiversity effects of improved forest management that comply to SFM standard requirements.*

IMPACTS OF CONVENTIONAL MANAGEMENT	RESIDUAL MSA	LOSS in MSA	AREA tropics	temperate	(10e6 ha) boreal	LOSS x AREA tropics	temperate	(10e6 ha*MSA)			
Selective logging	70%	-30%	18	39	-	-5,5	-11,6	-	-17		
Rotation forestry	50%	-50%	-	97	172	-	-48,3	-85,8	-134		
Plantations	30%	-70%	5	58	-	-3,2	-40,6	-	-44		
			23	193	172	-8,7	-100,6	-85,8	-195	-50%	
POTENTIAL CERTIFICATION IMPACTS	POSITIVE IMPACTS		AREA		(10e6 ha)	IMPACT x AREA		(10e6 ha*MSA)			
	min	max	tropics	temperate	boreal	tropics	temperate	boreal			
1. RIL application	10%	20%	18	39	-	2,7	5,8	-	8,5		on-site improvement
2. Better rotation forestry	10%	20%	-	97	172	-	14,5	25,7	40		on-site improvement
3. Plantation (reforestation effect)	20%	20%	2	29	-	0,5	5,8	-	6		restoration potential
4. HCVA set-aside in plantations	5%	40%	1	7	-	0,2	1,3	-	2		on site avoided loss
5. Avoided deforestation - sparing						*not quantified*			-		regional avoided loss
						3,4	27,4	25,7	57	29%	total potential effect

a forested area of 380 million ha, most of which is due to rotation forestry in the temperate region. This means that about 50% of the original biodiversity is lost when conventional forest management does not take biodiversity friendly practices into account.

The lower half of the Table presents the positive effects of different practices in SFM. The different effects of on-site improvement deliver the largest positive effects (measures 1 and 2), due to the extensive areas where rotation forestry is practiced. 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 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 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 almost 60 million ha·MSA. So implementing measures for improved forest management is potentially able to avoid about a third of the loss from conventional logging (60/195=31%). The last effect of sparing forest from deforestation cannot be calculated here, as it is not a direct result of certifying forest concessions. An indication can be obtained by performing a full land-use change scenario in a broader spatial and governance context.

Level 3 Continued: examples of supporting case studies

There are several case studies available that show that impacts of certified forest management, at different levels and in different contexts. Such case studies can for instance be found in van Kuijk et al (2009) and in the meta-analysis of Putz et al. (2012). The in-depth analysis by van Kuijk et al. (2009) concluded that, in spite of the large variety in responses between species, the different forest management practices associated with forest certification appear to benefit biodiversity in managed forests. But they clearly state that conclusions are tentative, as only a small number of well-designed impact studies were available.

Case studies on RIL management

A study on a tropical forest in Malaysia concluded that populations of endangered animals increased due to effects of certified forest management (Lagan et al. 2007).

In this FSC certified lowland forest in Malaysian Borneo (Sabah), RIL measures and HCVF protection were put in place to implement good forest management practices. By comparing general sources to the monitoring data of the Deramakot forest , the authors concluded that certified forest sustained denser populations of endangered large animals such as orang utans and elephants than elsewhere in Sabah. Other studies in the same region showed biodiversity benefits of RIL techniques on plants and soil macrofauna. For instance, tree species diversity was similar in an old-growth forest and a certified forest practicing RIL techniques, but was much lower in a forest where harvesting was practiced using conventional techniques. Next to species effects, effects on tree structure were also observed. In the RIL managed forest, canopy trees regenerated well, whereas in the conventionally harvested forest, pioneer tree species dominated the regeneration phase. Although this study was no strictly experimental but comparative, the presented evidence supports the positive claims of RIL on forest biodiversity.

Value of production forests for species conservation

It is often claimed that sustainably managed production forests can be complementary for species conservation to protected forests. An important driver of the loss of habitat for great apes is deforestation by (often) illegal logging This e iconic species are found in many FSC-managed tropical forest, although they are susceptible to human disturbance (van Kreveld & Roerhorst 2009). Viable populations of great apes depend on sufficiently large areas with suitable environments for them. In some tropical countries where primates occur there are hardly protected forest. Therefore, it is good to have a network of both protected areas and responsibly managed commercial forests. The authors emphasize that production forests must be responsibly managed, and preferably certified. The exact role that FSC-certified management plays is not yet clear. The positive effects are probably due to the use of selective logging with limited damage, protecting specific species and food sources, banning the use of forests for poaching, regulating hunting, and limiting access to.

The authors referred to a detailed study that presents unique density and distribution data of orangutans (Husson et al. 2008). In that study, little differences were found in orangutan numbers between areas that were not logged and those that were selectively logged. In conventionally logged areas, however, fewer orangutans were found. When adverse effects of selective logging were found, these turned out to be indirect (e.g., increased hunting via logging roads). The authors mention that the Borneo orangutan can better withstand the direct effects of logging than its Sumatran counterpart, probably because the Borneo orangutan is less specialized in its feeding habits.

Forest protection or Forest exploitation - Use it or loose it?

Avoiding deforestation is for a large part a matter of finance. An important result of forest certification is that it can give more value to a forest, making forest conservation economically relevant. A comparison between a protected forest and an FSC production forests in Mexico showed that generating income from sustainable logging is a good alternative to protection, for which finances are often difficult to find (Hughell & 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, provided that there is a financing source.

Conclusion

The case studies show that to create for biodiversity benefits it is wise to stimulate implementation of responsible harvesting techniques. Another important conclusion is that sustainably managed forests have an important conservation function that is complementary to strictly protection. Putz et al (2012) conclude “that selectively logged forests retain substantial amounts of biodiversity, carbon, and timber stocks, and therefore, this “middle way” between deforestation and total protection deserves more attention from researchers, conservation organizations, and policy-makers. Improvements in forest management are now likely if synergies among different initiatives can be enhanced, like retaining forest carbon stocks (REDD+), assuring the legality of forest products, certifying responsible management, and devolving control over forests to empowered local communities.”

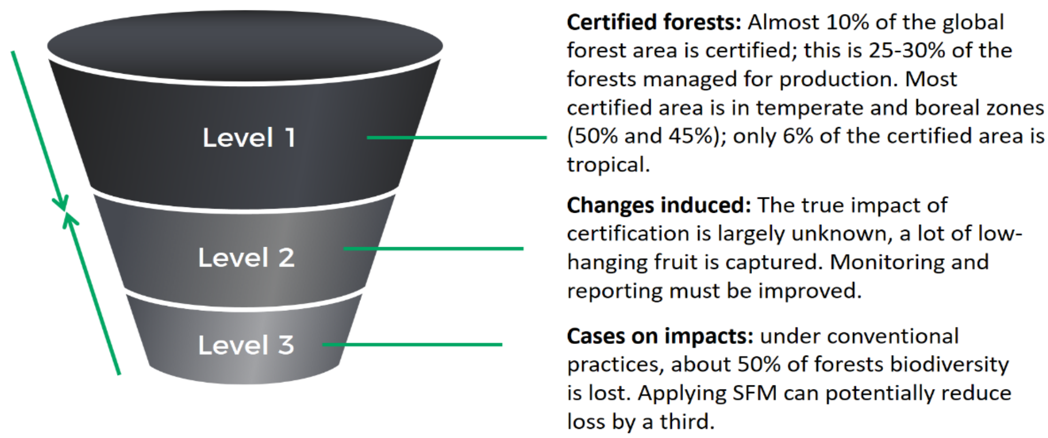


Figure 6.9 *Funnel analysis of forest biodiversity effects of SFM certification. Certified areas are well monitored, and knowledge om impacts is available from experimental cases. What is largely missing is monitoring of actual performance improvement.*

6.5. Expected future performance

For the assessment of the 4th Global Biodiversity Outlook, the secretariat for the CBD has published relatively simple statistical extrapolations towards 2020 for the area of certified forests, using trend data for FSC and PEFC up to 2014 (Leadley et al. 2014). From 2008-2013 the total certified area grew at an average annual rate of 6 per cent from 2008 to 2013. If this growth rate is applied on the 387 million ha. for 2014, forest certification may increase its area to about 470 Million hectare in 2020 (see *figure 6.10*), which would be an increase with 25%, compared to 2014/2015 numbers and a levelling of growth trends. This area would be 28% of 1.6 billion hectare of managed forest.

This extrapolation exercise does not take the underlying processes that drive these trends into account. It ignores that in certain regions of the world a saturation of the certified area has been reached. In the Nordic countries of the EU for instance, most of the production forests are already managed under some scheme of sustainable forest management. In the tropical areas where certification is still of marginal importance, upscaling of certification is confronted with several problems such as too high costs, that make certified wood production not a viable business (PWC &IDH, 2012).

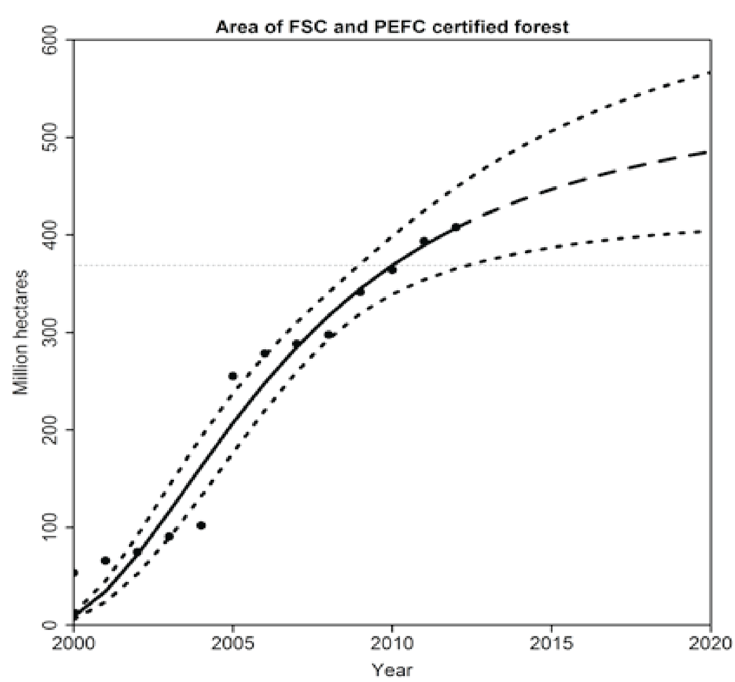


Figure 6.10 *Extrapolation of area of FSC and PEFC certified forest (source Leadley et al., 2014).*

Another issue is the potential for growth in certified forest plantations. Planted forests are projected to rise from about 260 to 300 million ha by 2020 (FAO 2010), but there are several concerns about their sustainability. For instance, the no-conversion principle means that plantations can only be established on abandoned agricultural lands (reforestation) or in degraded forests (restoration). This potential is uncertain, as the available land that meets these criteria is probably limited, and the costs of plantation establishment on these lands will be considerable. Establishing them in secondary forests is possible (depending on the local interpretation of " degraded forest"), but that might happen at the cost of biodiversity values.

The limitations point at the fact that certification is primarily a market based instrument, and only a means to an end, which is sustainable forest management. Within the certification world there is a clear realisation of the limited reach of the instrument and hence the need to look 'beyond certification', for instance by helping producer countries to make international standards part of their national forest laws. Still, recent Zero Net Deforestation Initiatives by the private sector may result in growing use of FSC/PEFC standards in the tropics to provide assurance for the no-deforestation claim (Mallet et al. 2016), see also Ludwig (forthcoming). To bring certification of semi-natural and secondary forests further in tropical regions, a combination of interventions and measures are necessary: cost reduction, certification premiums, ecosystem service payment, legal capacity building and integrated land-use planning.

6.7. Additionality to international biodiversity governance

In this section we zoom out again from certification schemes in forestry to the recognition of the contribution of all certification schemes to biodiversity targets (including also agricultural commodities and fisheries). The CBD Aichi target 7 clearly aims that "by 2020 areas under agriculture, aquaculture and forestry are managed sustainability, ensuring conservation of biodiversity".

Potential of certified production

In its evaluation of the state of progress towards achievement of the Aichi target 7 (Leadley et al. 2014), chapter 7 examines organic farming and conservation agriculture, FSC and PEFC, and aquaculture label MSC, as well as the community of practice for conservation agriculture.

The authors note that 'a wide range of commodity specific labelling and certification programmes exist' (like RSPO, RA, UtZ), but these are not all included in the evaluation of progress. They continue to note that 'many of these labels and certification programmes are used as a measure of sustainability', but question their relevance for sustainability given that the lack of consistent criteria that makes it difficult for consumers to judge their credibility, and raises doubts about their potential to yield measurable sustainability benefits.

So the question is whether it is a missed opportunity to not include more data on certified agricultural area such as those reported by ITC and IISD, and reflect within CBD context on policy recommendations such as those proposed by Potts et al., (2016).

A first comparison of area data shows that the contribution might be limited. Organic farming area (in 2014 43.7 million ha., 1% of global agricultural land) is about 4 times larger than certified sustainable agricultural area, as reported in The World Organic Agriculture 2016 (Willer & Lernouds 2016). In comparison, conservation agriculture (including practices like zero-tillage) is approx. 160 million ha in 2015, almost 4% of agricultural land.

But many relevant developments may not have been recognised sufficiently in judging the additionality of agricultural VSSs to the CBD targets. Potts et al. (2016) expect based on current market trends and existing "unimplemented" corporate commitments to sustainable sourcing, that standard-compliant production for each of the eight agro-commodities markets will have reached 10 per cent or more of total global production by 2020. This still means that certification of standard-compliant production only accounts for a small portion of total global agricultural land area with minimal presence in major staple crops.

Beyond certification

We have showed the potential of VSSs to contribute to at least conserving and possibly enhancing (by reforestation) biodiversity in resource production areas. Their value lies in an additional contribution to biodiversity conservation; complementary to strict ecosystem protection. In a future where enough resources will have to be produced for a growing world population, this approach and strategy becomes more important.

But it cannot be expected that full certification of agro-production will take place. This is partly due to the fact that certification is primarily a voluntary market based instrument, depending on demand by conscious (at present Western) consumes and the willingness of resource demanding supply-chain actors like producers and retailers to use VSSs in their company policies on CSR. And there are more obstacles to upscaling and mainstreaming certification. Transaction costs are considerable, and there is also a credibility crisis. Meeting the doubts and increasing the impacts will probably increase costs further.

Therefore, we need to look beyond certification and the uptake of consumer logos. Still, standards still have a lot to offer in terms of stakeholder consensus, infrastructure for control and assurance, market trust, producer benefits, and government support through procurement and sometimes uptake in national regulation of responsible land-use. So we expect that VSSs will still play a meaningful role in the near future, only in other forms and shapes than at present.

7 Conclusions and reflections

7.1. Introduction

This report is about the impact of so-called 'International Cooperative Initiatives on Biodiversity' (ICIBs). Five cases were analysed (alphabetical order): (1) Citizens' initiatives (CIs) that contribute to private nature conservation; (2) Community forest management (CFM) that strives for the improvement of rural livelihoods, forest conditions and forest biodiversity; (3) Landscape and forest restoration (LFR) under the Bonn challenge; (4) Re-naturing cities (RNC) for green infrastructures and biodiversity enhancement in urban areas; and (5) Voluntary Sustainability Standards (VSS) which – through market certification schemes – aims at enhancing the sustainable use of natural resources. All case studies were introduced in previous chapters, by describing their motives, goals, targets and theories of change (through the input-output-outcome-impact or I-O-O-I scheme). The core of the chapters, though, was the assessment of outcome and impact of the five ICIBs. In order to do so, we developed the 'funnel-shaped assessment framework', based on the work of Milder et al. (2015), that exposes three levels: (a) projects and areas realized ('outcome'), (b) positive biodiversity effects realized in terms of size of sustainable use areas, in terms of 'mean species abundance' (MSA) or in terms of both ('overall impact'), and (c) examples of positive biodiversity effects on the ground ('detailed impact'). Besides, indirect impacts were also traced in some of the case studies, realized through influencing policies of governments or management approaches of nature conservation organizations. For all cases, additionality (compared to governmental initiatives) and overlap (among the cases and with governmental initiatives) were discussed. Some cases also offer insights into the future, based on scenario analyses.

While writing this synthesis Chapter, the idea emerged to expand the concept of additionality, from solely a quantitative mode (additional hectares of sustainable use areas initiated by ICIBs that contribute to the conservation or enhancement of biodiversity, compared to protected areas initiated by governments) to also including a qualitative mode (increase of biodiversity – expressed as growth of MSA – in ICIB-initiated sustainable use areas, compared to 'conventional' natural resource management in comparable areas).

This final chapter will go into, first of all, summaries of the case study findings, secondly, into a synthesis of those findings, particularly with regard to the three case studies on forests (community management, restoration and certification), and thirdly into methodological reflection. The latter topic is particularly relevant for this study, because it is exploratory in nature, which led to some different methods and techniques tried-out in the various case studies.

7.2. Findings from the case studies

7.2.1 Citizens' initiatives (CIs)

Citizens and local communities have a long history of engagements with and contributions to the conservation of biodiversity in a range of green areas. This involvement of citizens in protection, maintenance and enhancement of biodiversity is not primarily driven by government policies, but by intrinsic motivation of citizens. Usually, CIs not only aim at benefits for biodiversity conservation, but also at co-benefits, such as developing accessible green spaces and at increasing environmental awareness. Because of the scattered and unorganised nature of CIs, only limited outcome and impact data are available. The current analysis is based on empirical results from only one country, the Netherlands (Mattijssen et al. 2016). The outcomes of the calculation are presented with high margins due to an high level of uncertainty.

In total, CIs are involved in nature management and restoration in an area that amounts to 0.3% to 14% of the total designated protected areas in the Netherland ('outcome'). Estimated impact is however significantly lower, between 0,1% and 1% of designated areas, due to the fact that many initiatives do not directly contribute to the conservation or enhancement of biodiversity (but to recreation or awareness raising, for example). Nonetheless, because of a lack of (reliable) data, possible impacts from political influence groups is not included in this calculation. Consequently, the actual contribution of CI's is expected to be significantly higher than the figures suggest in the above. The impact of green CIs is predominantly additional to efforts by governments or NGOs and are real impacts on the ground, not merely policy plans or bids that still need to be implemented. However, extrapolation from the Dutch context to an European, let alone the global context, is difficult, because emergence and success of CIs strongly depend on the environmental, social and governance context in each country.

While the main conclusion of this chapter may be that the direct impact of green CIs seems rather limited, indirect impact may outweigh these direct impacts. This includes pressure on governmental actors to improve biodiversity protection and contribution to the implementation of governmental and NGO policies through increased environmental awareness and public support for biodiversity conservation. Although these indirect impacts of CIs cannot be calculated, they seem relevant for all countries across the globe.

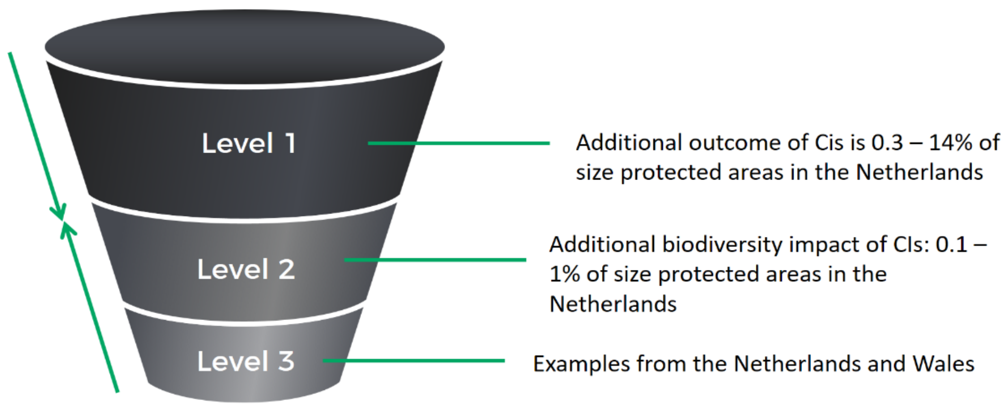


Figure 7.1 *Biodiversity impacts of CIs*

7.2.2 Community Forest Management (CFM)

CFM has become an influential approach in the management of forests around the world the last couple of decades. The central idea behind CFM is that local management of forests, either by communities or jointly with forest departments, is more effective than management by central state institutions alone, because local people are more involved and dependent on forests than far-away bureaucracies, thus better securing their conservation and sustainable use. This is particularly the case in Tropical developing countries, where forest-dependent livelihoods are widespread and where state institutions are often weak, corrupt or even absent. Already in the early 1970s, the idea of community participation was practiced in a few countries, advocated by NGOs and scientists and intensively discussed in the FAO at global level. Today, the CFM approach has diffused worldwide, including policy programs in many countries, covering about 360 million Ha., nearly 10% of the world's forests (see *figure 7.2* below).

Overall, CFM exhibits two goals: (1) to enhance the sustainable management of community forests; and (2) to improve forest-related livelihoods for local people. CFM's theory of change suggests that once forests are owned and/or (co)managed by communities, they will feel much more responsible for the resource, so that deforestation and forest degradation will be fought against, reduced, or even halted. We in this report particularly focus on CFM's first objective of enhancing sustainable forest management and on the latter's positive impact on forest biodiversity on the ground. Based on data from two datasets (FAO, 2015; RRI, 2014) and from three meta-analyses of CFM initiatives around the world (Arts & De Koning, 2017; Bowler et al, 2012; Persha et al, 2011), we found that about 115-135 million hectares of CFM areas contribute to biodiversity conservation and enhancement. In concrete terms, this often means an increase of tree density, an increase of species abundance, a return of certain lost forest species and the restoration of forest habitats.

We re-calculated these numbers into additionality figures for CFM today, compared to: (a) total forest protected areas (FPAs), (b) total terrestrial protected areas (tPAs) and (c) mean species abundance (MSA) in conventionally managed forests (see *figure 7.2* below). These figures amount to 18-21% (additional to FPAs), 5-6% (additional to tPAs) and 25% (additional MSA compared to conventional forestry), respectively. All in all, we conclude that CFM substantially contributes to forest biodiversity conservation (protection, sustainable use, avoided loss) and biodiversity enhancement (forest expansion, enrichment) at a worldwide scale. And if these trends continue over time, we might expect CFM areas with positive biodiversity impact to expand to about 180 and 240 million hectares in 2020 and 2030 respectively (based on a business-as-usual scenario).

Potential overlap among the cases and with governmental initiatives put these conclusions into perspective. Overlap with forest certification is low (about 5%), but with forest restoration unknown. The same applies to overlap with government initiatives on CFM. This percentage is probably much higher, but we do not have the data to calculate this number.

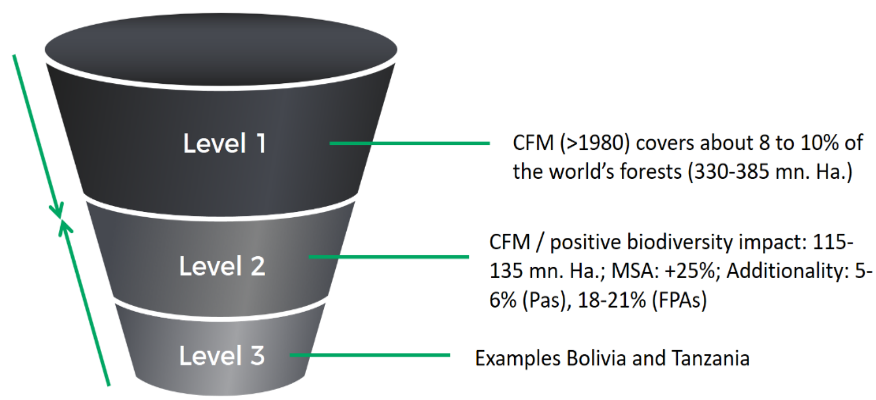


Figure 7.2 *Biodiversity impacts of CFM*

7.2.3. Landscape and forest restoration (LFR)

The Bonn Challenge specifically aims to offer practical means to tackle global warming, biodiversity loss and land degradation through forest and landscape restoration. The current analysis is based on the restoration commitments and pledges under the Bonn Challenge, domestic restoration plans of countries where data are available, restoration potential as calculated by the Global Partnership on Forest and Landscape Restoration (GPFLR) and positive impact on biodiversity with Mean Species Abundance (MSA) as indicator.
Restoration of landscapes on the ground ranges from small scale, bottom-up initiatives such as those in the Sahel, to large scale, top-down government-driven restoration programmes, such as in China. In the last few decades, there has been increasing international attention for landscape restoration, heavily driven by non-state actors. Restoring ecosystems and its functions with a landscape approach offers a holistic approach for multi-functional landscapes, in which conservation and restoration efforts are in balance with other land uses and human well-being.

The Bonn Challenge is to realize 150 million ha. of forests and land to be restored in 2020, and 350 million in 2030. This is about 7 and 16% of the 2.2 billion ha. restoration opportunity worldwide, as estimated by the GPFLR, respectively. To date, 136 million hectares are currently pledged to be restored in 39 different commitments (Jan. 16, 2017). This is about 6% of the 2.2 billion ha. restoration opportunity. And, outside of the Bonn Challenge, many countries have existing domestic targets for restoring degraded and deforested lands. These domestic targets amount to nearly 200 million ha. (or about 9% of the 2.2 billion ha of land suitable for restoration), with an average MSA increase of 26%.

However, all this does not mean that lands will be actually restored by 2020. These numbers are current pledges, and indicate a political commitment to start doing so. In that sense, the actual impact that the 136.32 million ha. pledged has on biodiversity will still depend on: (1) translation of this pledge into actual programmes and projects, (2) the enabling environment for these programmes to be successfully implemented, and (3) the type of restoration efforts that will be done on the ground in view of desired function improvement.

Figure 7.3 below summarizes the conclusions of the analysis. The Bonn Challenge was not specifically designed for the conservation of biodiversity, but has nonetheless a positive impact on biodiversity through restoration of natural habitats, both as direct effect or by-product of restoration practices and landscape approaches implemented under the Bonn Challenge.

As the Bonn Challenge's first milestone is to be realized in 2020, many countries are still in the process of designing National Action Plans. However, the case in Brazil does indicate that ICIBs might serve as crucial factors in going from legislation and National Programmes to actual implementation by pushing for crucial enabling conditions needed for on-the-ground action for sustainable conservation and protection. The success of the Brazilian Atlantic Forest Restoration Pact, however, is more an exception than a rule. In contrast, the role of ICIBs in the Democratic Republic of Congo (DRC) is much more limited, and they mainly serve as 'watchdogs' to keep restoration on the national political agenda. Consequently, as the actual implementation of the pledges is still ongoing, it is very difficult to assess actual outcomes, let alone impacts, to be expected from the Bonn challenge.

7.2.4. Re-naturing cities (RNC)

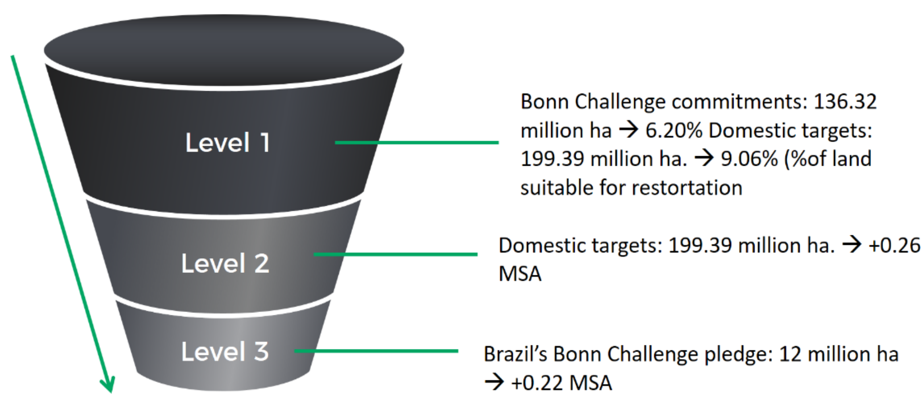


Figure 7.3 *Biodiversity impacts of LFR*

The concept of renaturing cities, from the perspective of local governments, offers possibilities for biodiversity conservation beyond national government efforts. As urban expansion frequently leads to habitat loss with severe consequences for biodiversity, renaturing cities provides a potential for restoring biodiversity and avoiding biodiversity loss resulting from ongoing urban development. City governments, municipalities and districts can act within the powers delegated to them by national governments, which in many cases allow them to go beyond national policy. The goals of a renaturing cities agenda can be manifold, but generally aim at maximising ecosystem services provided by urban green infrastructure, including biodiversity conservation.

Determining the contribution of renaturing cities by local governments to biodiversity conservation proves challenging. A key issue is the lack of benchmark data on urban biodiversity. This case study provides an indication of the number of cities that have biodiversity as an explicit policy goal. This is operationalized as cities having produced and published a dedicated biodiversity action plan, in the form of a Local Biodiversity Strategy and Action Plan (LBSAP), or similar document. The study shows a higher occurrence of such plans among larger cities. Of the world's 100 largest cities at least 12 had produced and published an LBSAP, or similar document. Extrapolation of data indicates that globally between 63 and 110 cities have urban biodiversity as an explicit policy goal, which constitutes around 4 to 7% of cities of over 0.3 million inhabitants worldwide.

Many of these urban biodiversity policies are of recent date, which suggests that it will take quite some time before adequate data are available for analysis of biodiversity impact. Perhaps the most promising way forward is provided by the City Biodiversity Index (CBI). This self-assessment tool can help cities monitor and evaluate biodiversity conservation and enhancement. Currently about 50 cities are in various stages of providing data for this index. Over time CBI benchmarks and follow-up assessments could facilitate analysis to determine the impact of urban biodiversity policies. With more data becoming available from ongoing projects and policies, more opportunities will arise for determining the contribution of urban biodiversity policies to nature conservation as part of renaturing cities agendas.

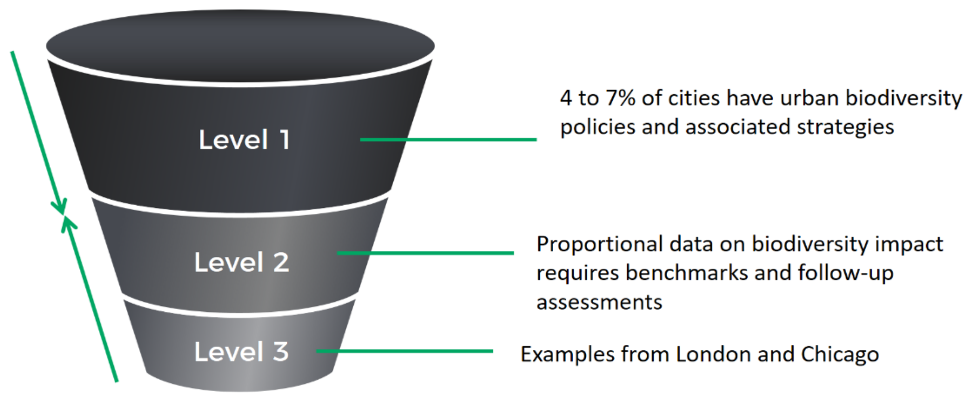


Figure 7.4 *Biodiversity impacts of RNC*

7.2.5. Voluntary Sustainability Standards (VSS)

VSS have become recognized mechanisms to connect consumption, production and trade, with the aim to create more sustainable development outcomes and impacts in value chains.

Over the last two decades, there has been a rapid increase in Voluntary Sustainability Standards (VSS) for agro-commodities, timber and fish. Starting early 1990s, these initiatives were originally initiated by environmental and social NGOs in industrialized countries, often in collaboration with market parties, aiming to raise awareness amongst conscious consumers to buy more sustainable products. They did this by setting standards for improved production, by working with local producers, and by introducing product labels to influence consumer choice.

Sustainability standards and certification have by now become more and more mainstream, with different actors entering the stage and new standards benefiting from (soft) infrastructure being in place. Over time these initiatives have been taken up by front-runners in business. Gradually, the type of commodities and number of products for which standards are set and implemented have grown. To show these improvements to consumers, a large number of product labels have been introduced. Also governments make use of market standards in their policies for procurement, giving another stimulus to market sustainability.

There is a generally observed lack of well-designed impact studies, both in terms of experimental design and focus on biodiversity. As a response to this critique, more and more impact studies have been conducted. Reviews on impact studies concluded that there is reasonable evidence that certification has had positive impacts on the environment and biodiversity in particular cases. However, the local variability in environmental conditions between sites will affect specific results, and together with methodological limitations, this does not allow simple extrapolation of the findings beyond the immediate cases under review. In impacts studies on agricultural standards for certification in coffee and cacao, much evidence was found suggesting that the inclusion of environmental criteria may indeed support biodiversity conservation, but it is less clear to what extent certification leads to improved conservation impacts.

We took a more detailed look at forest certification. For promoting sustainable management of forest (SFM) there are two main voluntary standards available, that exist already for more than 2 decades (FSC and PEFC). There has been a steady growth in the implementation of these standards in practice. Almost 10 % of the total global forest area is now certified, and this is 25-30% of the forest areas that is managed for production. Most certified area is found in the temperate and boreal zones (50% and 45%), only 6% of the certified area is tropical.

There is a lot of literature available on the effects of forest exploitation and SFM. Based on several literature sources and reviews, we estimated that under conventional forestry practices, about 50% of local forest biodiversity (MSA) will be lost. The application of different measures that are promoted under SFM can potentially reduce this loss by a third, bringing the loss back to about 35%.

The actual attained positive effects of SFM on forest biodiversity is not known. A lot of so-called 'low-hanging' fruit is captured under certification: standard forest operations that comply to SFM criteria are easiest and cheapest to certify. The real impact of certification is therefore still unknown due to insufficient reporting on changes in management and monitoring the biodiversity effects.

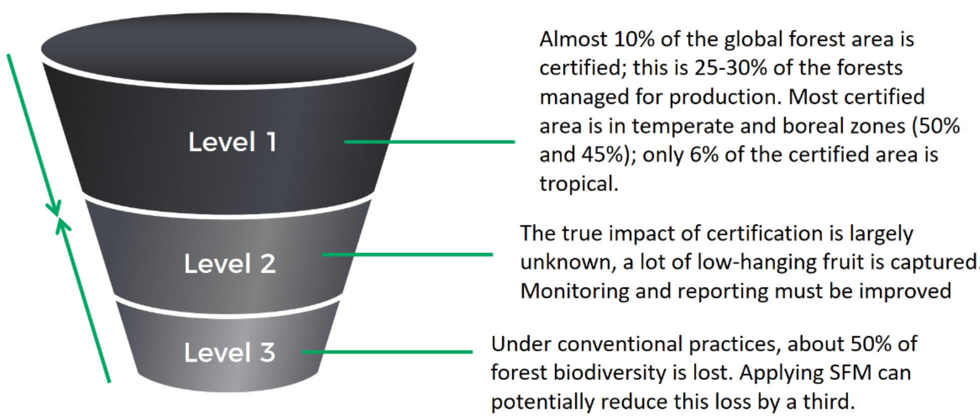


Figure 7.5 *Biodiversity impacts of forest certification*

7.3. Synthesis of findings

For the synthesis, we follow the three levels of the funnel-shaped assessment framework. Concerning level 1, the outcome of ICIBs, it should be noted first that not all cases produce complete and comparable outcome numbers, due to lack of data availability. For example, the 'renaturing cities' case only shows output data, hence the number of cities with urban biodiversity policies and strategies in place (4 to 7% of larger cities worldwide), without data to what extent these policies and strategies have been translated into actual and additional areas and qualities of green infrastructures in cities. These latter figures are unknown (so far). The same more or less applies to the 'landscape and forest restoration' case. Here we do know numbers with regard to areas and hectares, but these are only pledges and targets to be materialized in the next 5 to 15 years (about 135 million Ha. under the Bonn challenge and about 200 million Ha. under domestic targets, with an overlap of about 70 million Ha.). Here again, we can only talk about output (stated policy commitments), not outcome (attained behavioural change). The 'citizens green initiatives' case though comes one step closer to outcome figures, but the uncertainty of those are huge. New green areas realized by Dutch citizens' initiatives to date are estimated to fall within the (enormous) range of about 10.000 to 100.000 Ha. Moreover, this is a typically Dutch case that cannot be generalized to international levels. The remaining two cases – CFM and VSS (FSC/PEFC) – produce numbers closest to outcome data at global scale, respectively about 360 million Ha. (average of the range 330-385 million Ha.) and 390 million Ha. realized in practice. Together, CFM and forest certification thus cover a forest area of nearly 750 million Ha. (taking into account an overlap of certified community forestry of about 5 million Ha.). Compared to the current size of 'Forest Protected Areas' (FPAs) of 650 million Ha. worldwide (Morales-Hidalgo, 2015), this figure is impressive (although some overlap probably exists, because the IUCN definition of protected area status allows for limited use of natural resources in the less strictly-protected classes). So if we talk about the protection and enhancement of forest biodiversity today, it is as much about the sustainable use of forest areas as about the conservation of forests in national parks and nature reserves, at least in terms of size (750 versus 650 million Ha.)

The latter additive ('at least in terms of size') is crucial, because the realization of areas of CFM and forest certification does not necessarily guarantee that biodiversity is indeed better protected or enhanced through sustainable use. After all, in the scholarly literature, numerous examples of 'paper parks', 'CFM failures' and 'low-hanging fruit certification' are known, so one should dig deeper than area data. Now we move to level 2 of the funnel-shaped assessment framework: the impact of ICIBs (or the part of sustainable use initiatives or areas realized that actually contributes to biodiversity protection or enhancement). Unfortunately, for impact data to retrieve, one needs access to outcome data in the first place. As we have seen in the above, these are not known for all cases. So it is no surprise that we – besides outcome numbers – also lack impact figures for the 'renaturing cities' and 'landscape and forest restoration' cases. On top of that, impact figures of Dutch citizens initiatives are (again) quite uncertain and probably fairly underestimated, whereas those for forest certification are lacking. So only one case remains, CFM, for which several meta-studies are available in the scholarly literature from which impact data could be retrieved. This literature shows that about 35% of CFM initiatives (hence, about 125 million Ha.) actually produces positive biodiversity impacts. If this benchmark would also be valid for VSS (FSC/PEFC), which is still a wild guess, then about 260 million Ha. of forests managed through CFM and forest certification perform well in terms of biodiversity impact. Taking FPAs as benchmark (650 million Ha.), such would produce an 'additionality' of sustainable use areas in which positive biodiversity impact is realized of about 40%.

While executing our study, we also decided to add a qualitative biodiversity indicator, besides quantitative ones (surface, hectares), at impact level 2. Building upon earlier work of the Netherlands Environmental Assessment Agency (PBL), we adopted the MSA (Mean Species Abundance) indicator as a proxy for biodiversity, first for the VSS case, later for the CFM and LFR cases as well (note that biodiversity and MSA are not similar units, but for the sake of simplicity, we take them as synonyms in this report; see Chapters 3, 4 and 6 for further explanation;). MSA figures give an indication of the increase or decrease of mean species abundance due to changes in management practices of ecosystems or due to restoration efforts. Taken the forest cases together (CFM, LFR, VSS), ICIBs have realized an increase of biodiversity with about 15 to 25%. Such is possibly achieved in roughly 260 million Ha. of forests (see above).

Besides direct impact, ICIBs can also perform through indirect impact. They can influence governments, international organizations and/or nature conservation organizations to protect more biodiversity or protect it better. This type of impact is particularly relevant for citizens' initiatives, but potentially important for forestry community organizations, restoration initiatives, cities and certification bodies as well. This type of impact, however, is difficult to trace, let alone quantitatively to assess. Such figures therefore lack in this study. But the link between ICIBs and governments is more extensive than 'indirect impact' only. For example, for realizing new nature areas, formalization by governmental authorities is always a crucial precondition for effectiveness, since they might own the land, or design and monitor regulations for land use. In other words, working with governments is generally normal business for ICIBs. Besides, governments often become part of ICIBs over time; so the initiative is private, but for obvious reasons, governments become involved through supporting, facilitating and regulating roles (Mattijssen et al. Accepted).

Finally we move to level 3 of the funnel, with specific cases showing biodiversity impact. These cannot be synthesized by its very nature of being individual case studies. These nonetheless make explicit and tangible what the general figures from the above synthesis indicates (about 20% increase of biodiversity in roughly 260 million Ha. of forests due to ICIBs). We refer the reader to the individual chapters of this report to learn more about these specific examples.

7.4. Methodological reflection

Inspired by Milder et al. (2015), who provided a comprehensive framework for integrating different types of impact studies, we designed the 'funnel-shaped assessment framework' (see Figure 1.5, p. 5). We believe this framework was very helpful in grasping the impact of ICIBs. At the same time, we are also a bit disappointed in what we can really conclude on level 2 impact, the most interesting layer of the funnel from the perspective of the research aim of this study. Finding data for level 1 and exemplary cases for level 3 was not very difficult for all topics in this report, but level 2 particularly was. Generalizing biodiversity impact from the individual case studies or downscaling outcome data to impact figures turned out to be very challenging. For only one topic (CFM) we found data in the literature to downscale level 1 numbers to level 2 figures. And even in this case, conclusions come with substantial uncertainty. So our method (as any) is very sensitive to data availability, and obviously, the scholarly literature lacks impact data beyond multiple case studies or data from which overall impact can be retrieved (for as far as we could check the literature within the limited time frame of this study).

Besides, one can question whether a credible impact assessment, including inference of causal attribution, of a single intervention to an impact within a complex setting is possible (Ton et al., 2014). Maybe we should already be happy with some relevant outcome data and with some insights into the contribution of an intervention to a certain impact.

As a result of the lack of comparable data, the various studies in this report partially departed from one another in terms of methods and techniques, although all took the funnel as starting point. Initially, some focused on number of initiatives (CI, RNC), others on hectares (CFM, LFR), and one on biodiversity benefits itself (VSS). Later, combinations of approaches evolved (CI, CFM, VSS), the result of mutual learning from the various case studies. Consequently, the findings of the various chapters are not always easily to integrate (see the synthesis in the above).

Part of the variety in case studies can also be explained by the fact that we have been looking at different types of ICIBs, with clearly different 'lifetimes'. In hindsight, we conclude that we have considered at least two such types: (a) Type I: transnational mechanisms to facilitate conservation and sustainable use measures at lower levels of the spatial scale (LFR and VSS); and (b) Type II: lower level mechanisms to develop conservation and sustainable use measures at the local level (CIs and RNC). CFM is probably a hybrid form, in which transnational initiatives and local practices have been aligned in an 'glocal CFM movement' over time, based on and evaluated through internationally-sanctioned norms and criteria.

Type-I ICIBs can be typified as top-down. To a certain extent, it resembles more traditional, government-led biodiversity conservation approaches, in which higher scale institutions agree on aims and methods, after which efforts towards implementation are trickled-down. Although Type-I approaches in this report are initiated by non-governmental bodies, outcomes and effects highly depend on the successful implementation of these higher scale goals by both state and non-state actors. Methodologically, measuring output for these Type-I initiatives is generally not very difficult. However, quantifying the next steps in the implementation chain (see Chapter 1) – outcome and impact – is much more complicated. Measuring such effects critically depends on available data, methods and techniques.

Conventional measurements of the effects of type-I ICIBs generally focus on output: how many hectares are pledged under this regime, how many countries or companies are party to it, how many projects are planned, etc. Methodological challenges to move from output to outcome and impact particularly relate to the assessment of the implementation success rate of projects and the attribution of such success to the Type-I initiative concerned.

Type-II ICIBs can be typified as bottom-up, or better, bottom-linked approaches (Pradel, Garcia, and Eizaguirre 2013). These are practices at the lower level of scale, ranging from local communities (CIs) to cities (RNC). Focus of these practices is on direct outcomes or impacts. CIs often want to protect or enhance one single green area, or municipalities want to enhance green space in their city. Although networks may exist with other CIs or municipalities, the focus of Type-II ICIB is usually on local or regional impact, not on upscaling towards higher levels of scale. In addition, they tend to be more informal, with more focus on outcome and effect than on output. As such, they do not fit well into the implementation chain as presented in chapter 1.

Type-II ICIBs do generally not start with input from institutions on higher levels of scale, but with a diversity of local or regional practices. As a consequence, the methodological challenges to quantify the impact of Type-II ICIBs to national or supra-national scales are of different nature than the ones from Type-I ICIBs. For type II, the main challenges come from upscaling the diversity of outcomes and impacts to higher scale levels. Often, single case studies exist on impact of individual cases (Mattijssen et al. Accepted). However, the diversity of practices limits the possibility to reliably upscale and generalize these outcomes or impacts. In addition, and more than Type-I ICIBs generally do, Type II may also produce indirect effects, related to ICIB's engagements in governance systems in which civic and private actors complement governments (Biggs et al. 2012, Buijs et al. 2016). Challenges to improve estimations of overall Type-II effects therefore relate to the upscaling and generalizing of the diversity of local practices to higher levels of scale.

For next steps in research on ICIBs' impact, the following recommendations seem valuable. Firstly, it may be useful to consider the different types of ICIBs, with different pathways towards (potential) impact. The characteristics of each type come with specific challenges with regard to conceptualizing impact as well as with regards to methods and techniques to assess impact. Secondly, the scholarly literature should be further screened for meta-studies on and impact data about ICIBs, since the timeframe of this study was too limited for doing a literature search in-depth. Thirdly, impact data could be collected by a new research project itself, although such would probably require a full-fletched, million-dollars program. And finally, the funnel-shaped assessment framework could be further fine-tuned in terms of criteria, indicators and procedural steps.

7.5 Conclusion

First of all, this study shows – within all its limitations – that ICIBs are very relevant for biodiversity conservation and its sustainable use, besides (inter)governmental initiatives and those of 'classical' nature conservation organizations, particularly focusing on national parks and nature reserves. The report estimates – under certain assumptions and with substantial uncertainty – that ICIBs' additionality in forest biodiversity impact, for example, amounts to about 40% in quantity (size of sustainable use areas with positive biodiversity impact, compared to formally protected forest areas) and about 20% in quality (increase in biodiversity/ MSA compared to conventional forest management). These numbers exclude indirect impact of ICIBs, realized through governments or nature conservation organizations, because we were unable to quantify this effect (although we are convinced that indirect impact is substantial too). In other words, 'land sharing' in sustainable use initiatives is as important as 'land sparing' for nature conservation and biodiversity enhancement. Secondly, this study has used and adjusted an interesting framework – 'the funnel' – to assess the outcome and impact of ICIBs. However, this framework remains highly sensitive to specific data availability and needs further streamlining in terms of criteria, indicators and procedures. For that reason, it neither offers a fully standardized approach for individual studies yet nor can results from various studies be easily synthesized, as this report shows. Follow-up research should therefore focus on fine-tuning the framework and on further screening the literature for more meta-studies and impact data, or it should itself execute field data collection on biodiversity impact of ICIBs.

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Annex 1 – World maps and table overview of Bonn Challenge pledges (on top) and existing domestic restoration targets (bottom)

	Country	Bonn Challenge pledge (in million ha)	Existing domestic restoration targets (in million ha)
Bonn Challenge pledge > Domestic Targets	Costa Rica	1.00	0.23
	Cote d'Ivoire	5.00	2.11
	Ethiopia	15.00	14.30
	Ghana	2.00	1.67
	Guatemala	1.20	0.83
	India	21.00	10.40
	Kenya	5.10	4.21
	Peru	3.20	1.79
	Republic of the Congo	2.00	1.00
	Rwanda	2.00	1.59
Bonn Challenge pledge < Domestic Targets	Brazil	12.00	22.88
	Chile	0.50	0.6
	Colombia	1.00	2.02
	DR Congo	8.00	16.78 (+0.20)
	Mexico	7.50	10.48
	Pakistan	0.38	1.76
	Uganda	2.50	2.88
Bonn Challenge, no Domestic Targets	Argentina	1.00	
	Asia Pulp & Paper	1.00	
	Benin	0.50	
	Brazil's Atlantic Forest Restoration Pact	1.00	
	Burundi	2.00	
	Central African Republic	3.50	
	Ecuador	0.50	
	Guatemala Private Natural Reserves	0.04	
	Guinea	2.00	
	Honduras	1.00	
	Liberia	1.00	
	Madagascar	4.00	
	Malawi	4.50	
	Mexico (Campeche)	0.40	
	Mexico (Quintana Roo)	0.30	
	Mexico (Yucatan)	0.25	
	Mozambique	1.00	
	Nicaragua	2.80	

	Niger	3.20	
	Panama	1.00	
	Unites States	15.00	
No Bonn Challenge, only Domestic Targets	Azerbaijan		1.22 (+0.24)
	Burkina Faso		1.22
	China		15.77
	Indonesia		28.87 (+0.42)
	Lao PDR		7.59
	Lebanon		0.08
	Nepal		0.70
	Nigeria		30.04
	Vietnam		17.25
	Zambia		0.12
Bonn Challenge pledge just as high as Domestic Targets	El Salvador	1.00	1.00
	Total	**136.32**	**199.39 (+0.86)**

From Bonn Challenge website, accessed 26 January 2017

Annex 2 – Overview of existing domestic restoration targets and a break down in land use changes

Country	Existing domestic restoration targets (in ha)		
Azerbaijan	1,218,028 (+238,700)	Planted forests and woodlots	714,528
		Silviculture	503,500
Brazil	22.879.800	Planted forests and woodlots	18,781,300 (+238,700)
		Agroforestry	4,070,000
		Watershed protection and erosion control	28,500
Burkina Faso	1,218,000	Planted forests and woodlots	50,000
		Silviculture	73,000
		Agroforestry	1,010,000
		Watershed protection and erosion control	85,000
Chile	600,000	Planted forests and woodlots	100,000
		Silviculture	400,000
		Agroforestry	100,000
China	15,771,700	Planted forests and woodlots	15,575,700
		Natural regeneration	12,600
		Silviculture	60,100
		Agroforestry	3,000
		Watershed protection and erosion control	120,300
Colombia	2,017,984	Planted forests and woodlots	1,017,984
		Natural regeneration	1,000,000[1]
Costa Rica	234,347	Planted forests and woodlots	72,132
		Natural regeneration	162,215
Cote d'Ivoire	2,105,500	Planted forests and woodlots	246,000
		Natural regeneration	60,000
		Silviculture	1,799,500
DR Congo	16,775,750 (+200,000)	Planted forests and woodlots	13,026,700
		Natural regeneration	73,050
		Silviculture	1,276,000
		Agroforestry	2,400,000 (+200,000)

[1] also may include agroforestry, improved fallow, mangroves, watershed protection

El Salvador	1,000,000	Agroforestry	1,000,000[2]
		Planted forests and woodlots	5,931,200
		Natural regeneration	3,871,100
Ethiopia	14,302,300	Silviculture	4,500,000
		Planted forests and woodlots	1,100,000
		Silviculture	541,200
Ghana	1,667,200	Agroforestry	26,000
		Planted forests and woodlots	244,128
		Natural regeneration	5,160
		Silviculture	294,225
		Agroforestry	262,763
		Mangrove restoration	10,000
Guatemala	825,026	Watershed protection and erosion control	8,750
		Planted forests and woodlots	300,000
		Natural regeneration	800,000
		Silviculture	5,500,000
		Agroforestry	3,000,000
		Improved fallow	600,000
		Mangrove restoration	100,000
India	10,400,000	Watershed protection and erosion control	100,000
		Planted forests and woodlots	16,565,990 (+200,000)
		Silviculture	6,235,000 (+200,000)
		Agroforestry	20,000 (+20,000)
		Mangrove restoration	40,000
Indonesia	28,874,990 (+420,000)	Watershed protection and erosion control	6,014,000
		Planted forests and woodlots	4,100,000
		Silviculture	10,000
Kenya	4,210,000	Agroforestry	100,000
		Planted forests and woodlots	46,850
		Natural regeneration	7,040,000
Lao PDR	7,586,850	Silviculture	500,000
		Planted forests and woodlots	69,605
Lebanon	80,105	Natural regeneration	10,500

[2] total includes an unknown number of hectares for mangroves, gallery forest, and watershed restoration, and this will be determined following the ROAM process

Mexico	10,475,077	Planted forests and woodlots	3,627,130
		Silviculture	6,797,947
		Mangrove restoration	10,000
		Watershed protection and erosion control	40,000
Nepal	703,572	Planted forests and woodlots	12,000
		Silviculture	691,572
Nigeria	30,036,539	Planted forests and woodlots	13,750,700
		Natural regeneration	389,800
		Silviculture	50,000
		Agroforestry	15,704,039
		Mangrove restoration	130,000
		Watershed protection and erosion control	12,000
Pakistan	1,755,982	Planted forests and woodlots	1,305,100
		Natural regeneration	324,682
		Silviculture	100,000
		Watershed protection and erosion control	26,200
Peru	1,788,000	Planted forests and woodlots	1,213,000
		Silviculture	9,000
		Agroforestry	566,000
Republic of the Congo	1,001,000	Planted forests and woodlots	1,000,000
		Mangrove restoration	1,000
Rwanda	1,585,030	Silviculture	3,000
		Agroforestry	1,582,030
Uganda	2,883,000	Planted forests and woodlots	2,138,000
		Silviculture	720,000
		Agroforestry	5,000
		Watershed protection and erosion control	20,000
Vietnam	17,252,354	Planted forests and woodlots	2,650,000
		Natural regeneration	750,000
		Silviculture	13,754,800
		Mangrove restoration	97,554
Zambia	116,700	Planted forests and woodlots	81,020
		Silviculture	15,000
		Agroforestry	20,680

		Planted forests and woodlots	103,719,067 (+438,700)
		Natural regeneration	14,499,107
		Silviculture	43,833,844 (+200,000)
		Agroforestry	29,869,512 (+220,000)
		Improved fallow	600,000
		Mangrove restoration	388,554
Total	**199,364,834** **(+858,700)**	**Watershed protection and erosion control**	6,454,750

From Bonn Challenge website, accessed 26 January 2017

Annex 3 - Steps towards successful implementation

The Bonn Challenge pledges are not a guarantee for the successful implementation of landscape restoration. To learn about the conditions that could enable this, we studied cases in Australia, Brazil, China, Costa Rica, Ethiopia, Indonesia, the Sahel, the US and Vietnam (Bennett, 2008; Brancalion et al., 2014; Brown et al., 2014; Buckingham, 2015; Buckingham et al., 2014; Burger, 2002; Calvo-Alvarado et al., 2009; Cao et al., 2009; Chokkalingam et al., 2001; de Jong, 2010; Hartshorn et al., 2005; Reuben and Buckingham, 2015; Robins, 2004; UNCCD, 2015). We found the following preconditions:

1. Political Momentum as an enabling condition for landscape restoration - Framing a common agenda is crucial in getting actors involved. Current political momentum for restoration is partly due to its linkages to domestic issues and the added value of restoration to meeting international targets. A leading role for local institutions is imperative for local actors to take ownership of and see value in restoration efforts. Informal, flexible systems with low barriers and participation costs help bring together such stakeholders (Wentink, 2015).

2. Safeguarding restoration quality - landscape restoration should take into account the local, natural ecosystem, be targeted through time, across landscapes, in prioritized key areas (Manning and Lindenmayer, 2009). Policies play an important role in safeguarding this.

3. Trade-offs are acknowledged and addressed - Restoration efforts are often a trade-off between goals and ecosystem services. It is key to address that such trade-offs exist and consider these early in the design and implementation process (Caspari et al 2014, draft).

4. Stakeholder involvement on different levels - Landscape and forest restoration efforts have been proven more successful when multiple stakeholders became active participants and rural development objectives were incorporated in program design. For instance, land users with leadership skills and knowledge of climate change and degradation issues can trigger the involvement of their peers in restoration activities (de Jong, 2010; Curran et al. 2012).

5. Multi-sector involvement – the 2008 economic crisis increased risk regulations for private financing and dried up much public funding. In addition, growing public awareness for the environment made companies worry about their reputations. As a result, restoration has increasingly become a business practice. Public-private partnerships (PPP) offer a strategy to include multiple actors, receive funding from multiple sectors, and aid knowledge sharing.

6. Supporting regulations & legislation - Legislation can support intrinsically motivated actors, if it fits current knowledge and practices. This works if restoration policies and legislation do not conflict with other policies, or they will undermine each other (perverse incentives).

7. Financial incentives - to cover investment, maintenance, monitoring and opportunity costs, international funding, public and private sector investments are critical. However, investing without guarantee of project longevity and returns is risky. Local business cases help to decrease transaction costs and risks, while improving the likelihood of returns (Sewell et al., 2016).

8. Available and accessible information - systems to disseminate information on monitoring and implementation provide an way to share learnings and enhance political momentum.

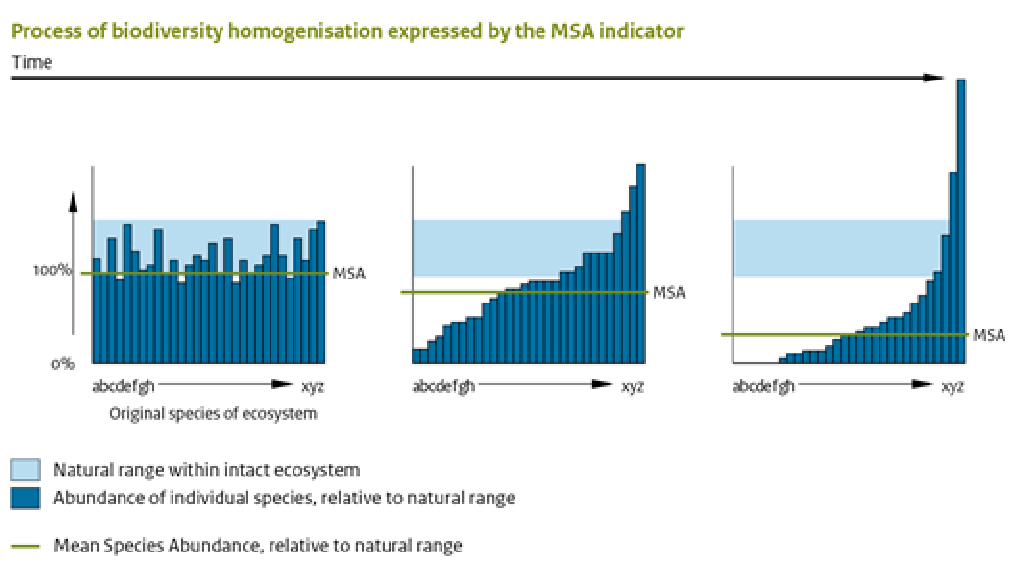
Annex 4 - The MSA indicator and use in biodiversity calculations (adapted from (Kok et al. 2014)

Mean Species Abundance (MSA)

MSA is used in the impact calculations as the biodiversity indicator, with the consequence that the focus is on preserving naturalness, ecosystem intactness and species abundance than, for instance, on species richness. The MSA indicator considers the variety of plant and animal species in a certain area and their population sizes (number of individuals or abundance). The various nature types or biomes in the world vary greatly in the number of species, their species composition and their species abundance. Obviously a tropical rainforest is entirely different from tundra or tidal mudflats. The process of biodiversity loss as a result of human activities is at first characterised by the decrease in abundance of many original species and the increase in abundance of a few other -opportunistic- species. As a result, many different ecosystem types are becoming more and more alike, the so-called homogenisation process (see the *figures in A4.1* below showing this process from left to right). Decreasing populations are as much a signal of biodiversity loss as highly expanding species, which may sometimes even become plagues in terms of invasions and infestations.

Biodiversity loss is calculated in terms of the mean species abundance (MSA) of the original species compared to the natural or low-impacted state. This baseline, the species composition and abundance of the original ecosystem, is used here as a means of comparing different model and calculation outputs, rather than as an absolute measure of biodiversity. If the indicator is 100%, the biodiversity is assumed to be similar to the undisturbed or low-impacted state, implying that the abundance of all species equals the natural state. If the indicator is 50%, the average abundance of the original species is 50% of the natural or low-impacted state and so on. To avoid masking, significant increased populations of original species are truncated at 100%, although they should actually have a negative score. Exotic or invasive species are by definition not part of the calculations, but their impact is taken into accounted by the decrease in the abundance of the original species they replace. The MSA at global and regional levels is the sum of the underlying biome values, in which each square kilometre of every biome is equally weighted. The regional or global MSA is determined by multiplying the impact of different pressures and summing the MSA values of different use types and ecosystems. This calculation method is visualised in *figure A4.2.* For more information on MSA and the relationship with environmental pressures, see Alkemade et al. (2009) and www.globio.info.

Figure A4.1: *Biodiversity loss is characterized by a decrease in abundance of original species and the increase in abundance of a few, often opportunistic, species as a result of human interventions. Extinction of species (left hand side of the graph on the right, species a – f) is the last step in the homogenization process, changes in species abundance is an early warning signal that precedes actual extinction.*



How to interpret MSA changes? Global MSA is used throughout this report as an overall indicator of the impact of a certain option. As with any aggregated index, changes in values may be difficult to interpret. The changes in MSA values occur because of changes in environmental pressure and the extent of ecosystems. Changes in the values can thus also be expressed in both indicators. The reference MSA value for 2010 of 68% implies that globally 32% of the original naturalness of ecosystems has disappeared. However, a considerable part (24 %) of the global (remaining) MSA is tundra and desert systems, biomes types that are difficult to convert. The total historical loss of 32% is equivalent to a loss of the size of Asia in terms of its biodiversity value. Similarly, future trends can be evaluated. The baseline shows an additional MSA loss of 9 percent points, equivalent to a loss of the size of North America in terms of biodiversity value. The loss is almost exclusively forest and grassland ecosystems, with little change in desert and tundra systems.

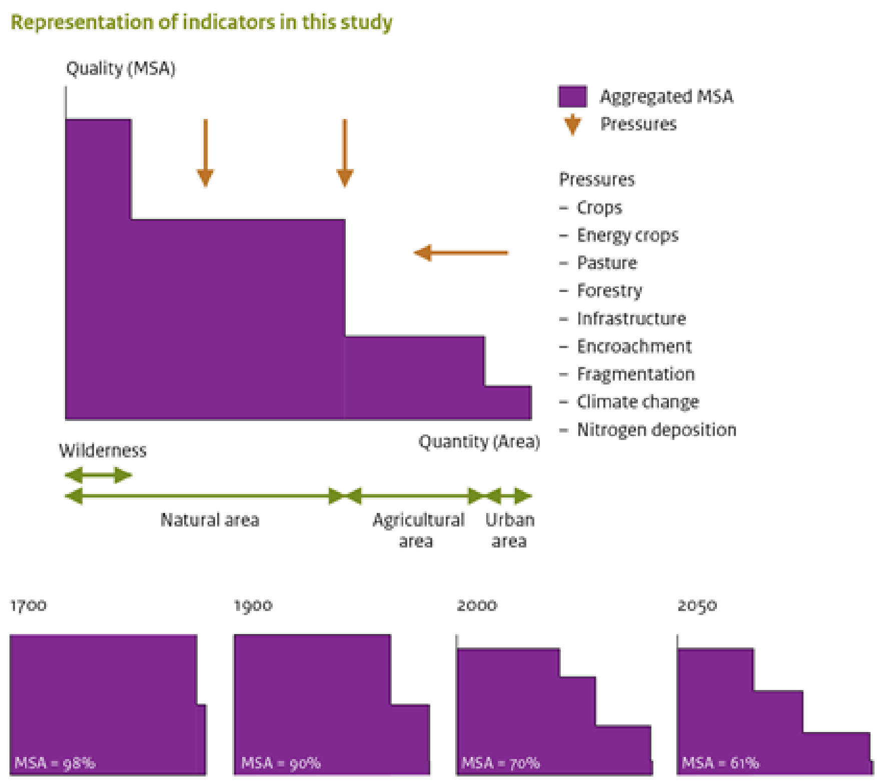


Figure A4.2 *The MSA methodology. Ecosystems have two characteristics: a quantity measure (area) and a quality measure (MSA index). For both aspects, the original state is used as reference and equals 100% by definition. Pressures include land-use changes to agriculture, livestock breeding and forestry. Climate change also leads to MSA loss, but is not taken into account in this study. The trend from 1700 to 2050 is illustrated in the lower part of the figure. Real calculations at detailed grid level show greater variation in results than suggested here.*

2017