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COMPENSATING THE BIODIVERSITY IMPACTS OF LAND USE: TOWARD ECOLOGICALLY EQUAL EXCHANGE IN THE NORTH–SOUTH CONTEXT

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Cover image by Michael Curran

Dedication

To Mirjam, the Vorder Hüttli, its inhabitants and my family—just for being there.

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"There is something fundamentally wrong with treating the earth as if it were a business in liquidation."

– *Herman Daly.*

Abstract

Increasing consumption and global trade of agricultural products is driving rapid declines in biodiversity in agricultural frontier regions, primarily in the Global South. Rising demand for food, fodder and biofuel in developed and emerging nations is driving displaced habitat and species impacts via trade, requiring a strengthening of consumer responsibility in the affluent Global North. Improved regulation and the strengthening of international treaties relating to biodiversity loss and environmental protection are required. At the same time, international compensation schemes for ecological damages, such as biodiversity offsets, represent one immediate option to address impacts. However, at least four methodological hurdles stand out: (1) whole-supply chain impact assessment tools, such as Life Cycle Assessment (LCA), must be improved to adequately integrate biodiversity loss; (2) the biodiversity offset methodology must be improved to reduce risks, adequately reflect ecological value and operate internationally; (3) different conservation strategies must be employed in the right context to ensure effective, fair and additional conservation gains; and (4) opaque international supply chains mean impact location is often unknown, thus impact assessment and compensation must be assessed under spatial uncertainty. This dissertation attempts to resolve some of these issues, and in the process, investigate the potential for an international application of the biodiversity offset methodology in order to compensate producing countries of the Global South for consumption-induced biodiversity loss. In the following sections, *Chapter 1* provides an overview of the issues, reviewing the literature on consumption, trade, biodiversity loss and ecological compensation. Economic justifications for conserving biodiversity are also reviewed, along with novel policy tools that have emerged in recent years to meet this goal.

Chapter 2 focuses on the impact assessment step, (hurdle 1), and reviews approaches to model biodiversity loss in Life Cycle Assessment (LCA). The chapter illustrates conceptual and methodological shortcomings of the LCA framework. Different impact categories assess loss at different scale (e.g. global warming assesses global losses of species, whereas land use assesses local and regional losses). This reduces the relevance of comparisons between impact categories and the results of single-score aggregation techniques. There is also poor taxonomic and geographic coverage. Only three of the five main drivers of biodiversity loss are represented (identified by the Millennium Ecosystem Assessment). The chapter concludes with a review of assessment techniques from ecology and conservation to help LCA researchers fill these gaps.

Chapter 3 focuses on methodological approaches to compensation (hurdle 2), critically testing the ecological validity of "biodiversity offsets". Offsets assume that the restoration of biodiversity

is timely and predictable, justifying the use of habitat restoration to compensate impacts on old growth ecosystems. In order to ensure equitable biodiversity offsets, three criteria must be met: (i) diversity in secondary and old growth habitat must converge over time, (ii) active restoration must accelerate the process, and (iii) current offset policies must account for predicted uncertainties and time lags using appropriate "offset multipliers". To test if these criteria are met, *Chapter 3* presents a meta-study of the restoration and secondary habitat literature. Using data from 108 studies comparing diversity in secondary and old growth habitat, statistical models of the age-diversity relationship are constructed and used to predict "recovery times" under passive and active restoration. The results indicate that full recovery of species composition may take centuries or more. Active restoration significantly accelerates the process. However, considering the uncertainties and risks involved, offset ratios are required that exceed what is currently applied in practice.

Chapter 4 focuses on conservation implementation (hurdle 3), and assesses the cost-effectiveness of two direct conservation strategies: payments for ecosystem services (PES) and land purchases or easements (LPE) in Central Kenya. Conservation costs are modelled across a case-study landscape using survey and literature data. Management costs are collected from four regional conservation organizations and use in a site selection process for conservation interventions (PES or LPE) over 30 years, favouring areas of low cost and high ecological value. The LPE strategy results in larger and less fragmented reserves, representing about 50% more species than PES. However, the legal and policy context in Kenya presents challenges to wider uptake of land purchases or voluntary easements. Because of simpler institutional requirements, PES represent a more immediate property-based conservation tool. However, where possible, conservation organizations should aim for long-term, once-off investments to secure habitat at the lowest possible cost.

Chapter 5 returns to the issue of impact assessment (hurdle 1), and develops a new assessment method for land use in LCA[1]. The method is based on an index of "threat and rarity-weighted" species richness associated with different land use classes. The method is compared to two previous LCA approaches (based on local relative change in species richness, and SAR-based regional changes, respectively) using a cases study of export crops in East Africa. It demonstrates how modelled agricultural data on yield and production volumes can help localize impacts and reduce uncertainties (hurdle 4). The results show how the biodiversity impacts of different crops (tea, coffee and tobacco) vary mainly based on where they are produced. Recommendations are put forward for sourcing products or locating future agricultural land in areas that minimize risks to global mammal diversity.

Chapter 6, brings together the previous work to develop a conceptual framework for the international application of biodiversity offsets. The chapter investigates application of the framework to a hypothetical case study of improved conservation in Central Kenya (using results from *Chapter 4*) compensating land use impacts at multiple scales in East Africa depending on knowledge on the location of production (using the method from *Chapter 5*).

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[1] This chapter represents a contribution as co-author rather than lead author

The chapter applies the same biodiversity index to quantify land use impacts and conservation gains. Gains are corrected by applying "offset ratio multipliers" (based on *Chapter 3*) that translate uncertain, risky, and delayed conservation gains into discounted present values that can be compared to present impacts. Where the impact location is uncertain, compensation activities are targeted using a spatial conservation planning approach to align offsets with conservation priorities. Conservation cost and agricultural production data are used to assess likely compensation costs per unit area or agricultural product. The results indicate that "biodiversity neutral" production could be achieved with a conservation premium on farm-gate price of *ca.* 35%–280% (range of 25% and 75% quartiles of values), depending on the scale of impact assessment (sub-national, national, regional), costs included (e.g. pure conservation goals, public protected area support, development budget) and the type of impact compensated (occupation or transformation impacts). For final consumers in the Global North, price increases would likely be lower because production costs make up a smaller fraction of final prices (perhaps reducing premiums by a factor of 5–10). This range represents a ballpark estimate encompassing many simplifying assumptions and methodological choices. Further research is required to develop a consistent conceptual basis for valuing biodiversity (e.g. based on *ecocentric* or *anthropocentric* values), choosing an appropriate discount rate for these values, and dealing with fundamental uncertainties and data limitations. The ethical, institutional and social fairness dimensions of international offsets also need to be carefully considered before any such scheme is implemented.

Chapter 7 concludes the dissertation with a review of the main messages. A critical analysis of the research as a whole is presented, with reference to weak points and potential improvements. Finally, the practical implications of the work are discussed with respect to policy relevance and potential avenues for implementation. This includes an overview of conservation financing mechanisms (for collecting and dispersing funds internationally) and a brief review of the international policy context, such as commitments under the Convention on Biological Diversity and existing bilateral and multilateral trade agreements via the World Trade Organization.

Zusammenfassung

Steigender Konsum und globaler Handel von Agrargütern führen zu rapiden Verlusten an Biodiversität in landwirtschaftlichen Grenzgebieten, hauptsächlich im Globalen Süden. Zunehmender Bedarf an Nahrung, Futter und Biokraftstoffen in Industrie- und Schwellenländern treiben verlagerte Lebensraum- und Artenauswirkungen durch Handel an, was die Stärkung der Konsumentenverantwortung im wohlhabenden Globalen Norden erfordert. Verbesserte Regulierungen und die Stärkung von internationalen Abkommen mit Bezug auf Biodiversitätsverlust und Umweltschutz sind erforderlich. Gleichzeitig stellen internationale Ausgleichspläne für ökologische Schäden wie Ausgleichsmaßnahmen zum Erhalt der biologischen Vielfalt eine unmittelbare Option dar, um sich mit den Auswirkungen zu befassen. Jedoch stechen mindestens vier methodische Hürden hervor: (1) Methoden für die Wirkungsabschätzung vollständiger Lieferketten wie die Ökobilanz (Lebenszyklusanalyse (LCA)) müssen verbessert werden, um Biodiversitätsverluste hinreichend zu berücksichtigen; (2) Biodiversitätsausgleichsmaßnahmen müssen verbessert werden, um Risiken zu reduzieren, ökologische Werte widerzuspiegeln und international anwendbar zu sein; (3) verschiedene Umweltschutzstrategien müssen im richtigen Kontext angewandt werden, um effektive, gerechte und zusätzliche Umweltschutzerfolge zu gewährleisten; und (4) undurchsichtige internationale Lieferketten bedeuten, dass Standorte, an denen sich die Auswirkungen ereignen, oft unbekannt sind und deshalb Wirkungsabschätzungen und Ausgleiche unter räumlichen Unsicherheiten bewertet werden müssen. Diese Dissertation versucht einige dieser Probleme zu lösen und währenddessen das Potenzial für eine internationale Anwendung von Biodiversitätsausgleichsmaßnahmen zu untersuchen, um Herstellernationen des Global Südens für durch Konsum ausgelöste Biodiversitätsverluste zu kompensieren. In den folgenden Abschnitten liefert das Kapitel 1 einen Überblick der Fragestellungen, wobei Literatur über Konsum, Handel, Biodiversitätsverluste und ökologische Ausgleiche besprochen wird. Ökonomische Rechtfertigungen für den Erhalt biologische Vielfalt werden auch überprüft, zusammen mit neuartigen politischen Instrumenten, welche sich in den letzten Jahren herausgebildet haben, um dieses Ziel zu erreichen.

Kapitel 2 konzentriert sich auf die Wirkungsabschätzung (Hürde 1) und es werden Ansätze zur Modellierung von Biodiversitätsverlusten in Lebenszyklusanalysen (LCA) überprüft. Das Kapitel zeigt konzeptionelle und methodische Unzulänglichkeiten innerhalb der LCA-Rahmenstruktur auf. Verschiedene Wirkungskategorien bewerten Verluste unterschiedlicher Skalen (z.B. schätzt globale Erwärmung globale Artenverluste ab, während Landnutzung lokale und regionale Verluste abschätzt). Dies reduziert die Bedeutung von Vergleichen zwischen

Wirkungskategorien und Ergebnissen von Einzelwertaggregationstechniken. Auch die taxonomische und geographische Abdeckung ist schwach. Nur drei der fünf Haupteinflussfaktoren von Biodiversitätsverlusten werden dargestellt (identifiziert in der Millenniumsstudie zur Bewertung von Ökosystemen). Das Kapitel schließt mit einem Überblick über Bewertungstechniken aus der Ökologie und dem Umweltschutz, um LCA-Forschern zu helfen diese Lücken zu füllen.

Kapitel 3 konzentriert sich auf methodische Ansätze zum Ausgleich (Hürde 2), wobei die ökologische Validität von Biodiversitätsausgleichsmaßnahmen kritisch überprüft wird. Ausgleichsmaßnahmen nehmen an, dass die Wiederherstellung der biologischen Vielfalt schnell und vorhersagbar ist, was die Nutzung von Lebensraumrenaturierungen zur Kompensierung von Auswirkungen auf unberührte Ökosysteme rechtfertigt. Um angemessene Biodiversitätsausgleichsmaßnahmen sicherzustellen, müssen drei Kriterien erfüllt werden: (i) die Vielfalt in Sekundär- und Primärlebensräumen muss mit der Zeit konvergieren, (ii) aktive Renaturierung muss den Prozess beschleunigen und (iii) derzeitige Ausgleichsinstrumente müssen vorhergesagte Unsicherheiten und zeitliche Verzögerungen berücksichtigen, indem geeignete Ausgleichsmultiplikatoren verwendet werden. Um zu überprüfen, ob diese drei Kriterien erfüllt wurden, legt Kapitel 3 eine Metastudie über Literatur über Renaturierung und Sekundärlebensräume dar. Indem Daten aus 108 Studien verwendet werden, in denen die Vielfalt in sekundären und ursprünglichen Lebensräumen verglichen wird, werden statische Modelle der Alter-Vielfalt-Beziehungen erstellt und genutzt, um „Erholungszeiten“ unter passiver und aktiver Renaturierung vorherzusagen. Die Ergebnisse weisen darauf hin, dass eine vollständige Erholung der Artenzusammensetzung Jahrhunderte oder länger dauern kann. Aktive Renaturierung beschleunigt den Prozess erheblich. Werden jedoch die damit verbundenen Unsicherheiten und Risiken berücksichtigt, werden Ausgleichsraten benötigt, die das übertreffen, was derzeit in der Praxis angewandt wird.

Kapitel 4 konzentriert sich auf die Umsetzung des Umweltschutzes (Hürde 3) und bewertet die Kosteneffizienz zweier direkter Umweltschutzstrategien: Zahlungen für Ökosystemdienstleistungen (engl. payments for ecosystem services (PES)) und Landkauf und Nutzungsrecht (engl. land purchase and easements (LPE)) in Zentralkenia. Umweltschutzkosten werden innerhalb einer Fallstudienlandschaft modelliert, wobei Daten aus Umfragen und der Literatur verwendet werden. Verwaltungskosten werden von vier regionalen Umweltschutzorganisationen gesammelt und in einem Prozess zur Standortwahl für Umweltschutzeingriffe (PES oder LPE) über 30 Jahre verwendet, wobei Gebiete mit geringen Kosten und hohen ökologischen Werten bevorzugt werden. Die LPE-Strategie hat größere und weniger fragmentierte Schutzgebiete zur Folge, welche ungefähr 50% mehr Arten vertreten als PES. Jedoch stellen die rechtlichen und politischen Rahmenbedingungen in Kenia Herausforderungen für eine stärkere Verbreitung von Landkäufen und freiwilligen Nutzungsrechten dar. Aufgrund der einfacheren institutionellen Anforderungen stellt PES ein direkteres auf Eigentum basiertes Umweltschutzinstrument dar. Wo möglich sollten Umweltschutzorganisationen jedoch langfristige, einmalige Investitionen anstreben, um Lebensraum zu den kleinstmöglichen Kosten zu sichern.

Kapitel 5 kehrt zum Thema Wirkungsabschätzung zurück (Hürde 1) und entwickelt eine neue Bewertungsmethode für Landnutzung in LCA. Die Methode basiert auf einer Kennzahl für „bedrohungs- und seltenheitsgewichtete“ Artenvielfalt, die mit unterschiedlichen Landnutzungsklassen in Zusammenhang steht. Die Methode wird mit zwei früheren LCA-Ansätzen verglichen (basierend auf lokalen relativen Änderungen der Artenvielfalt bzw. SAR-basierte regionale Änderungen), wobei eine Fallstudie von Nutzpflanzen für den Export in Ostafrika verwendet wird. Es demonstriert, wie modellierte agrarwirtschaftliche Daten zu Ertrag und Produktionsvolumen helfen können, Auswirkungen zu lokalisieren und Unsicherheiten zu reduzieren (Hürde 4). Die Ergebnisse zeigen, wie Auswirkungen verschiedener Nutzpflanzen (Tee, Kaffee und Tabak) auf die Biodiversität hauptsächlich aufgrund dessen variieren, wo sie produziert werden. Es werden Empfehlungen zur Beschaffung von Produkten oder Lokalisierung von zukünftigen Ackerflächen in Gebieten, die die Risiken auf die globale Säugetiervielfalt minimieren, vorgebracht.

Kapitel 6 bringt die vorangehende Arbeit zusammen, um eine konzeptionelle Rahmenstruktur zur internationalen Anwendung von Biodiversitätsausgleichsmaßnahmen zu entwickeln. Das Kapitel untersucht die Anwendung der Rahmenstruktur auf eine hypothetische Fallstudie von verbessertem Umweltschutz in Zentralkenia (unter Verwendung von Ergebnissen aus Kapitel 4), wobei Auswirkungen der Landnutzung auf mehreren Skalen in Ostafrika in Abhängigkeit von der Kenntnis des Produktionsstandortes kompensiert werden (unter Verwendung der Methode aus Kapitel 5). Das Kapitel wendet denselben Biodiversitätsindex an, um Auswirkungen der Landnutzung und Umweltschutzerfolge zu quantifizieren. Erfolge werden korrigiert, indem „Ausgleichsmultiplikatoren“ angewandt werden (basierend auf Kapitel 3), welche unsichere, riskante und verzögerte Umweltschutzerfolge in abgezinste Barwerte übersetzen, die mit gegenwärtigen Auswirkungen verglichen werden können. Wo der Standort der Auswirkungen ungewiss ist, wird auf Ausgleichsaktivitäten gezielt, wobei ein räumlicher Umweltschutzplanungsansatz verwendet wird, um Ausgleichsmaßnahmen mit Umweltschutzprioritäten in Einklang zu bringen. Daten zu Umweltschutzkosten und landwirtschaftlicher Produktion werden verwendet, um die wahrscheinlichen Ausgleichskosten pro Flächeneinheit oder Agrargut abzuschätzen. Die Ergebnisse weisen darauf hin, dass „biodiversitätsneutrale“ Produktion mit einem Umweltschutzaufschlag auf den Ab-Hof-Preis von ca. 35%–280% (Spannweite der 25% und 75% Quartile der Werte) in Abhängigkeit der Skala der Wirkungsabschätzung (sub-national, national, regional), den inbegriffenen Kosten (z.B. reine Umweltschutzziele, öffentliche Förderung eines Schutzgebietes, Entwicklungshaushalt) und der Art kompensierter Auswirkung (Auswirkungen durch Besetzung oder Umwandlung) erreicht werden könnte. Für Endverbraucher im Globalen Norden würden die Preiserhöhungen wahrscheinlich niedriger sein, da die Produktionskosten einen kleineren Anteil der Endpreise bilden (möglicherweise Reduktion des Aufschlags um einen Faktor von 5–10). Diese Spannweite stellt eine grobe Abschätzung dar, welche viele vereinfachende Annahmen und methodische Entscheidungen umfasst. Weitere Forschung ist erforderlich, um eine konsistente konzeptionelle Grundlage zur Wertschätzung von Biodiversität zu entwickeln (z.B. basierend auf ökozentrischen oder anthropozentrischen Werten), einen angemessenen Abzinsungssatz für diese Werte zu wählen, und sich mit grundlegenden Unsicherheiten und beschränkter Datenverfügbarkeit zu befassen. Die ethischen, institutionellen und sozialen Gerechtigkeitsdimensionen der internationalen Ausgleichsmaßnahmen müssen zudem sorgfältig berücksichtigt werden, bevor irgendein Vorhaben umgesetzt wird.

Kapitel 7 schließt die Dissertation mit einem Überblick der Kernaussagen ab. Eine kritische Analyse der Forschung als Ganzes mit Bezug auf Schwachstellen und potenzielle Verbesserungen wird präsentiert. Letztendlich werden die praktischen Konsequenzen der Arbeit hinsichtlich politischer Relevanz und Umsetzungsmöglichkeiten diskutiert. Dies schließt einen Überblick über Umweltschutzfinanzierungsmechanismen (zur internationalen Sammlung und Verteilung von Fonds) und eine kurze Besprechung des internationalen politischen Rahmens wie der Verpflichtungserklärungen nach dem Übereinkommen über die biologische Vielfalt und existierenden bilateralen und multilateralen Handelsabkommen durch die Welthandelsorganisation ein.

Chapter 1

Introduction

1.1 Ecologically unequal exchange

The scale of the human economy has reached unprecedented proportions in recent decades. There is growing evidence that humanity is approaching, or has already exceeded, multiple environmental planetary boundaries in order to provide goods and services for a growing and more affluent society (Rockström *et al.* , 2009; Running, 2012). This has led to severe and accelerating reductions in biodiversity and ecosystem services globally (MA, 2005d; Butchart *et al.* , 2010; Nelson *et al.* , 2010). Ecosystem degradation caused by agricultural land use represent perhaps the greatest single threat to the maintenance of biodiversity and provisioning of ecosystem services (MA, 2005c; Balmford *et al.* , 2012). Human activity affects roughly 80% of the planet's terrestrial surface area (Ellis & Ramankutty, 2008) and *all* marine ecosystems (Halpern *et al.* , 2008). Currently, 12% of global ice-free land is devoted to cropland and a further 26% to pastureland (Foley *et al.* , 2011). Humanities "groundwater footprint" — the aquifer area required to supply current rates of groundwater use — is roughly 3.5 times larger than the total globally available aquifer area (Gleeson *et al.* , 2012). Overexploitation of surface water has also led to over 65% of all river discharge-dependent habitats being classified as threatened (Vörösmarty *et al.* , 2010). The alarming rate at which natural heritage is declining is testament to deep structural cracks in the human-environment relationship. Yet declines continue, despite repeated calls for frugality, precaution, environmental justice, social fairness, and intergenerational equity — the essential tenets of sustainable development.

Conflicts between environmental protection and satisfying global demand for agricultural products will intensify in the coming decades, as the human population swells to a predicted 9 billion by 2050, and consumption patterns in developing and emerging nations converge on western norms (Nelson *et al.* , 2010; Tilman *et al.* , 2011). It is estimated that roughly 1 billion additional hectares of agricultural land will be needed by 2050 to meet demand for a doubling of agricultural output (Tilman *et al.* , 2011; Foley *et al.* , 2011). The majority of future habitat conversions will take place in the developing and emerging nations of the "Global South", where remaining tracts of suitable land and water resources remain for exploitation. Production within current global agricultural value chains is already shifting to the Global South, largely to meet rising demand both in industrialized nations and in economies in transition (Weinzettel *et al.* , 2013; Fader *et al.* , 2013). The rapid rise in direct purchases or long-term leasing of huge tracks of land, so-called "land grabbing", is testament to a recent global rush for land and water resources (Tscharntke *et al.* , 2012a). In the last two decades of the 20th century, tropical forests in the "Global South" were the principal source of new agricultural land, and this trend is likely to continue into the near future (Gibbs *et al.* , 2010).

The spatial decoupling of consumption from production, and its associated ecological impacts, creates local environmental externalities for producing countries that are not compensated by final consumers (e.g. loss of biodiversity and other undervalued ecosystem services that are not reflected in product prices). This phenomenon is known as "ecologically unequal exchange" (Hornborg, 1998b; Røpke, 2001) and describes how industrialized countries use their advantageous position in the network of global exchange — gained through imbalances in accumulated wealth, technology, political and military strength — to secure highly favourable terms of trade. This facilitates disproportionate access to natural resources and the sink capacities of ecological systems (Rice, 2007a; Hornborg, 1998b). The perpetuation of ecologically unequal exchange is said to lead to so-called "ecological debt", which reflects an ongoing and cumulative use of natural resources and waste sinks (such as land, water, and the atmosphere) by the Global North, constricting present and future use by developing countries (Goeminne & Paredis, 2010).

Ecologically unequal exchange has been demonstrated in the context of trade between the Global North and South in a range of recent studies. Regarding land and water resource, Fader *et al.* (2013) highlighted unequal patterns in external dependence driven by trade. Weinzettel *et al.* (2013) illustrated a link between affluence and displaced land use. Shandra *et al.* (2009a; 2009b) demonstrated a strong link between both deforestation rates, and mammal extinction risk, with trade volumes of agricultural commodities and other primary products derived from land occupation and habitat transformation. Rice (2007a) quantified ecological footprint flows between nations, characterizing highly unequal patterns in the use of environmental space, with producer nations essentially supplying ecological goods and services at the expense of their own consumption. In a wide-ranging study, Srinivasan *et al.* (2008) valued these external costs in monetary terms. Across a range of drivers of environmental degradation, the study illustrates stark inequalities in the distribution costs and benefits of production (Srinivasan *et al.*

, 2008). Between 1961 and 2000, northern consumption of deforestation-related wood and agricultural products created global environmental externalities with a net present value of between \$36.3 and \$274 billion (international 2005 dollars). Of that total, only \$17 billion in costs were born directly by high-income countries (between 6.2% and 46% of the total global costs). For external costs driven by intensification and expansion of agriculture in already-converted and non-forested biomes, the figures were more equitable, where between 59% and 83% of the global external costs of northern consumption were born by northern consumers.

Biodiversity loss and international trade

Amongst the many ecological and social impacts related to international trade, recent research has revealed its major role in driving biodiversity loss. Instances of species and ecosystem threat have risen massively in recent years due to an increase in demand for many intensively produced commodities, such as palm oil, soya, beef, leather and paper (e.g. Koh & Wilcove, 2008; DeFries *et al.* , 2010; Rudel *et al.* , 2009; Lenzen *et al.* , 2012; Shandra *et al.* , 2009a,b). Lenzen *et al.* (2012) collected global data on over 25'000 instances of species risk linked to commodity production (e.g. pulp, paper, or palm oil production) and harvesting of wild animal and plant populations (e.g. fishing, logging). The study characterized the risk-intensity ("biodiversity footprint") of these products, and analysed its flow between countries using a multi-region input-output analysis (MR–IOA). The findings indicate that while developed countries generally possessed relatively low levels of species threat, and low net exports of biodiversity-threatening products, they are the main culprits in importing and consuming high-impact commodities. Effectively, this amounts to exporting biodiversity loss to the Global South.

Allocating responsibility for these losses along the supply chain from producer to supplier to consumer is not necessarily straightforward. Producers can control the conditions of production and what they sell (thus "enabling" downstream damages to occur; Lenzen & Murray, 2010), but have to adapt to the preferences of downstream suppliers. While suppliers can decide which producers to buy from, and can decide whom to sell to, they also respond to the demands of consumers/households. Consumers decide what they buy and consume, but also have an impact on upstream impacts by deciding where they sell their labour (Lenzen & Murray, 2010). All parties therefore are be expected to bear some degree of responsibility for impacts along the entire supply chain, as all parties profit (i.e. "shared responsibility" Lenzen *et al.* , 2007).

This perspective of shared responsibility assumes that the freedom to choose exists for all parties, that imbalances in power do not exist, that information is perfect, and that the effects of distortions and externalities are minimal (e.g. monopolies, technological lock-in, budget constraints, unpriced servies). In reality, these factors are prolific, and actors are not always "sovereign" in their decision-making ability. This will influence terms of trade, which will not always result in mutual benefits for both parties. Ecologically unequal exchange and ecological debt (see above) represent a manifestation of the imbalances in *ecological* benefits/impacts between parties (Hornborg, 1998b). This perspective challenges the assumption of sovereignty along the value chain, and views trade between industrialized and developing nations as a direct expression of power relations, wealth imbalances, and historical circumstances (Hornborg, 1998b; Jorgenson & Clark, 2009). Responsibility for both the prevention and compensation of damages is seen to disproportionately fall on the consumer. In reality, both viewpoints are correct to some degree, and a fair allocation of responsibility is likely to be complex and system-specific. Recent debates relating to climate change mitigation, emissions rights, and embedded greenhouse gases in traded commodities reflect this complexity (Lenzen *et al.* , 2012; Lenzen & Murray, 2010). Despite this, the concept of "full consumer responsibility" has gained acceptance regarding compensation of carbon emissions, especially among consumers and suppliers in the industrialized world (e.g. voluntary carbon offsets, carbon net-neutrality). In this context, a similar application to the issue of consumption-driven biodiversity loss appears justified (Lenzen *et al.* , 2012).

1.2 Why does biodiversity loss matter?

Arguments to take biodiversity loss seriously at all levels of national and international policy making are often linked to the provisioning of services by intact ecosystems (ecosystem services; ES). These include carbon sequestration (contributing to climate regulation), water purification, the provisioning of economic goods, and cultural or traditional values associated with biodiversity (Wallace, 2007). This view was recently and extensively popularized by the Millennium Ecosystem Assessment (MA, 2005d), which highlighted an alarming and widespread loss of ecosystem diversity and functions worldwide. Biodiversity was recognized in the MA (2005d) as both the underlying foundation for many ecosystem services, and in terms of it's existence and non-use economic values. Monetary valuation of ES was recommended as an appropriate tool for communication purposes (including the use of Cost-Benefit Analysis as a policy tool for assessing the net benefits of development projects). The recent release of the "TEEB" report and section of books (The Economics of Ecosystems and Biodiversity; TEEB, 2011) has developed this monetization approach to biodiversity, inspired by the treatment of climate change impacts by the "Stern Report" (Stern, 2007). However, monetary valuation continues to be severely criticized, and suffers from high uncertainties associated with monetary estimates of non-use and intrinsic values. These uncertainties result from the choice of valuation methods, their technical details (e.g. influence of the payment vehicle, type of survey, hypothetical bias), the inadequate representation of deep cultural or ethical convictions, the assumption of commesurability of values, and poor knowledge of complex ecosystem dynamics (e.g. Spash, 2007; Frame & O'Connor, 2011; Atkinson *et al.* , 2012).

Alongside monetary valuation, an explicit focus on *biodiversity maintenance* has also been proposed as a key policy goal in itself (Blignaut & Aronson, 2008). This perspective highlight the extreme complexity of social-ecological systems, and our deep uncertainties regarding present and future costs and benefits of biodiversity loss. The concept of biodiversity maintenance is analogous to the application of the precautionary principle, the Safe Minimum Standard (SMS) (Berrens, 2001; Ciriacy-Wantrup, 1963; Seidl & Tisdell, 2001) or the "critical natural capital" framework (Ekins *et al.* , 2003). De Groot (1992) argued that biodiversity maintenance is "one of several regulating functions that keep ecosystems intact, self-organizing, and effectively adapting to change in an evolutionary sense", which was later echoed by Daily (1997). A policy of biodiversity maintenance emphasizes the critical role of biodiversity in the human-environment relationship, in terms of maintaining environmental resilience, redundancy and stability (sensu Holling, 1973). A large body of research illustrates the effect of high taxonomic diversity, intact functional groups and complete food-webs on reducing the likelihood of so-called "regime shifts" (Folke *et al.* , 2004a). Regime shifts represent abrupt changes in the functions, structure, identity and feedbacks of a social-ecological system, bringing it from one stable state to another (often undesirable) one (Kinzig *et al.* , 2006). Examples include rangeland transitions from grass to bush-dominated communities, due to the interaction of fire, herbivory and predation, or the collapse and reorganization of exploited fisheries (Carpenter *et al.* , 2011b).

In any case, there is an established recognition amongst almost all actors involved in biodiversity policy, of the increasing scarcity, marginal value (monetary and otherwise), and non-substitutability of biodiversity. This calls for stronger public intervention to address the overexploitation of a public good, through the deployment of existing and novel strategies to preserve the planets biological diversity (Helm & Hepburn, 2012).

1.3 Meeting the global costs of conservation with novel economic tools

Current international funding to prevent or slow the loss of species and ecosystems is woefully inadequate (Balmford *et al.* , 2003a). Improving and expanding the worlds system of protected areas to adequately represent species and ecosystems of conservation concern will likely to require anything up to a 10-fold increase in current funding, depending on the ecosystem type and targets (Balmford *et al.* , 2004; James *et al.* , 2001b; Balmford *et al.* , 2003a; Bruner *et al.* , 2004; McCarthy *et al.* , 2012) These figures do not include adaptation of protected areas in response to climate change-induced shifts in the ranges of species and ecosystems (Hagerman *et al.* , 2010). Currently, the developing world records both the highest current shortfalls and the highest future needs, and will require the majority of future funds (Balmford *et al.* , 2003a).

To mediate this transfer of both exiting and new funds, the Convention on Biological Diversity (CBD) mandates that developed countries are liable to bear the incremental costs of global biodiversity protection in meeting the 17% protection target set in the Nagoya protocol (which has been signed by 92 of the 193 countries that ratified the CBD). However, current funds available through the Global Environment Facility (GEF), the official mechanism of intergovernmental transfers to balance cross-border externalities (both positive and negative), fall well short of meeting this target (Oliver, 2004b,a). Alternative funding avenues for biodiversity conservation are thus clearly desirable, including new conservation mechanisms that go beyond the command-and-control approach of protected areas (Helm & Hepburn, 2012; Chazdon *et al.* , 2009b). A range of new economic mechanism have emerged in recent years to address the biodiversity problem (Landell-Mills & Porras, 2002; Helm & Hepburn, 2012; Miteva *et al.* , 2012). These can largely be split into “beneficiary pays” or “polluter pays” schemes (Engel *et al.* , 2008).

Beneficiary pays schemes aim at capturing a willingness to pay for a perceived service (including direct payments for biodiversity in terms of hectares of entire ecosystems or habitat for a specific species), by linking downstream beneficiaries of that service to upstream providers. Such schemes have been popularized by the recent emergence of diverse Payments for Ecosystem/Environmental Service (PES) schemes (Wunder, 2005; Engel *et al.* , 2008). Ecological services provided by biodiversity, along with its option, existence and bequest values are enjoyed globally, but particularly in the Global North, where the willingness to pay for conservation is also considered highest (Balmford & Whitten, 2003). Recent research has proposed deploying international PES schemes (IPES) to harness this willingness to pay, and provide a positive incentive for southern countries to “supply” ecosystems and their services rather than exploit them (Farley & Costanza, 2010; Farley *et al.* , 2010).

An alternative and complementary option is based around the *polluter pays* principle, and stipulates penalizing the economic drivers of biodiversity loss and degradation. A number of authors have linked degrading activities (including perverse subsidies) with conservation funding shortfalls (e.g. Myers, 2003; Balmford *et al.* , 2004; Barbier, 2012). Under such a scheme, a negative incentive is created to both reign in damaging economic activities, and provide much needed conservation funds. The fact that ecological debt and ecologically unequal exchanges largely act on a North-South axis (discussed above) implies that a polluter pays system would also channel funding in the same direction as a user pays system. The two schemes can thus be considered complementary.

Linking ecological debt to compensation mechanisms

The use of ecological compensation, in particular, has been proposed as one effective mechanism to capture the value of ecological services and biodiversity in international economic decision making (Darbi *et al.* , 2009). The application of compensation mechanisms in relation to ecological impacts (both voluntary and mandatory) has grown enormously in recent years, both in the developed and developing world. "Biodiversity offsets" represent perhaps the most prominent compensation mechanism, employed under different auspices in different regions. While ecological compensation may take the form of financial payments between parties, offsets are unique in that they emphasize scaling of impact to compensation based on biophysical units of measurement, such as vegetation quality and patch size (e.g. the "Habitat Hectares" approach towards scaling compensation; Parkes *et al.* , 2003).

Offsets operate through a combination of both positive and negative economic incentives. They increase the cost of development by applying the polluter pays principle, holding the developer liable for ecological damages. The requirement for the creation of similar ecological value elsewhere (or the purchase of biodiversity credits) creates a positive incentive for private actors to invest in the "supply" of biodiversity (Wissel & Wätzold, 2010). Offset policy may require a "like-for-like" recreation of impacted biodiversity, where substitutability between different forms of biodiversity is not permitted (i.e. *very strong* sustainability; Ekins *et al.* , 2003), or allow a "like-for-unlike" trade, where substitution between different types of biodiversity is permitted, (i.e. *strong* sustainability), as long as the agreed measure of "biodiversity value" is equivalent (ten Kate *et al.* , 2004). While elegant in practice, both empirical and theoretical research on offset policy continue to highlight numerous conceptual, institutional and practical failures, leading to calls for substantial changes to the model and implementation (Figure 1.1).

Biodiversity offsets in theory and practice

Biodiversity offset policies currently focus on mandating the compensation of biodiversity impacts of local development projects through "habitat trades" between an *impact site* and *offset site* (Figure 1.1, left pane). At the offset site, habitat is either created through habitat restoration, or loss is "averted" through improved protection (Maron *et al.* , 2012).

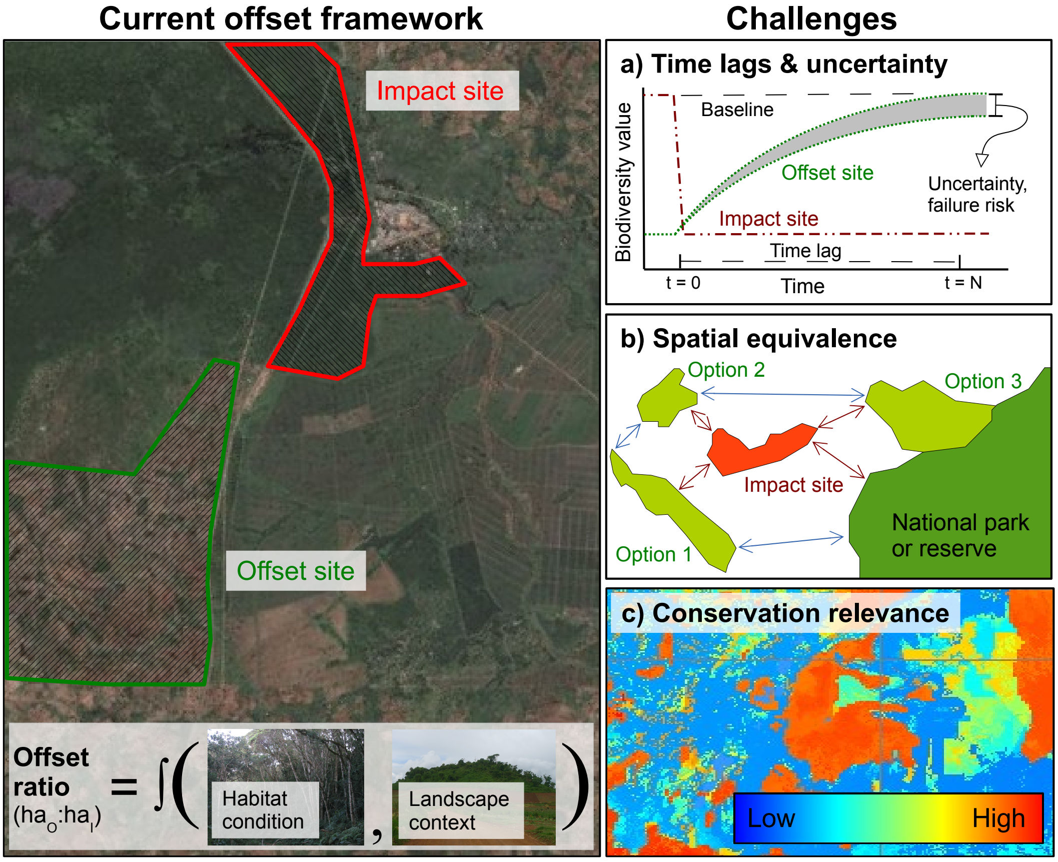

Figure 1.1 – Offset framework in terms of current status and major challenges. Current methods for calculating offset ratios, or for assessing the equivalence of habitat trades (left pane), generally focus on local factors. The “offset site” (green polygon) is compared to the “impact site” (red polygon) based on habitat area, condition and basic indicators of landscape context (e.g. native vegetation coverage in neighbourhood). Recent research has criticized the simplicity of this approach, and suggested changes (right pane). The main issues are (a) the time profile of offset transactions: time lags and uncertainties are high for restoration offsets due to the delayed delivery of biodiversity value; (b) the spatial equivalence and agglomeration of traded habitat patches: offset ratios should reflect spatial factors, such as the distance to neighbouring patches or agglomeration with existing reserves; (c) the integration of offsets and spatial conservation planning: offsets need to strengthen existing reserve networks and adapt to regional patterns of species and ecosystem diversity, rarity, threat etc. ha_{O} = hectares offset, ha_{I}=hectares impacted.

The ratio of biodiversity value gained to value lost is quantified using "offset ratios", based on an integrative habitat quality or service index (Parkes *et al.* , 2003; Strange *et al.* , 2002). For example, the "habitat hectares" approach (Parkes *et al.* , 2003) uses an integrative index of vegetation structure, anthropogenic disturbance, and neighbourhood landscape indicators, in a fixed weighing system to compare habitat patches of differing condition in a consistent way (but see McCarthy *et al.* , 2004 for a critique). This habitat quality comparison is used to convert gains and losses into simple habitat area ratios (i.e. hectares gained to hectares lost).

Offset ratios may, but usually do not, include additional corrections (multipliers) for time delays in the gain of biodiversity value, present and future uncertainties, the risk of restoration failure, and the spatial configuration of habitat in the landscape (Bruggeman *et al.* , 2005; Moilanen *et al.* , 2009a; Bekessy *et al.* , 2010). Recent reviews of biodiversity offset policies have thus been very critical of the lack of sound ecological theory and the (un-)fairness of exchanges in practice. Empirical research shows a poor record of compliance and monitoring (Bull *et al.* , 2012), high occurrence of restoration failure (Maron *et al.* , 2012), and little accounting of time lags (Moilanen *et al.* , 2009a) or spatial considerations (Bruggeman *et al.* , 2005). Areas proposed for improvement include (i) using "offset multipliers" based on info-gap theory to deal with uncertainty, (ii) the use of empirical/theoretical models to assess restoration failure risk, (iii) time discounting to correct for time delays in biodiversity delivery, (iv) embedding the offset concept within a spatial planning framework to take account of patch configurations, landscape dynamics and non-linearities, and (v) align offset gains with broader conservation priorities using spatial conservation planning tools (Overton *et al.* , 2013; Moilanen, 2012a).

Biodiversity compensation through offsets currently operates locally or regionally on a case-by-case basis. There have been calls for a global offset mechanism to compensate land use-driven biodiversity loss associated with heavily traded, high-impact agricultural commodities, such as beef, leather, soya, palm oil and wood products (Peterson *et al.* , 2008, 2011). Essentially, this scheme proposes addressing the problem of ecologically unequal exchange between North and South countries by channelling compensation through biophysical, rather than economic, repatriation. While the scheme focuses on compensating, rather than preventing, damages, it represents an embodiment of the consumer responsibility principle and is a necessary starting point in switching to biodiversity friendly production (e.g. through correcting price differences between ecologically damaging and less-damaging products).

While specifics are sparse, such a scheme *could* operate through a range of mechanisms: intergovernmental financial transfers (e.g. via the GEF), unilateral action by a single nation or club of nations using tariffs on imported/exported commodities, or a voluntary scheme funded by consumers, retailers or producers, and implemented through a form of international payments for biodiversity services (e.g. Victor, 2006; Farley & Costanza, 2010; Wuenscher & Engel, 2012). It is important to point out that an international PES-like scheme applies the PES mechanism only to implement payments, improve cost-effectiveness, and ensure continued supply. One characteristic feature of PES is the bargaining between buyer and seller over the amount of biodiversity supplied[1] (i.e. buyer preferences and willingness to pay determine levels biodiversity supplied). This feature is replaced in a compensation mechanism by fixed requirements balancing absolute measures of biodiversity impact and gain. Requirements are fairly determined through the appropriate use of offset ratios and multipliers.

1.4 Challenges and research gaps

On a practical level, designing an international compensation mechanism for biodiversity loss involves many hurdles, both conceptual and methodological. At a minimum, four major methodological and data related challenges can be identified that hinder implementation.

1.4.1 Missing impact assessment tools

Relatively coarse, sector-wide assessments of economic and trade data, such as input-output analysis or economy-wide Life Cycle Assessment (e.g. Shandra *et al.* , 2009b; Lenzen *et al.* , 2012; Rice, 2007a), are widely applicable, and can illuminate important but general patterns of impact distribution between nations and across sectors. However, supply chains that bring individual products from initial transformation of raw resources, through production to final consumption involve complex patterns of causality driving ecosystem degradation (e.g. an agricultural product is produced through land occupation in one country using inputs produced in another, is then transported to, and processed, in a third before being consumed in a fourth). At the same time, detailed assessments of causal relationships at small spatial scales (e.g. patterns of oil palm expansion into forests in South-East Asia; Koh, 2007a) are highly valuable, but not yet widely applicable.

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[1] It is worth pointing out that the many instances of PES that involve non-voluntary participation, little bargaining, or indirect interaction between buyer and seller has caused many authors (Muradian *et al.* , 2010; Kosoy & Corbera, 2010; Tacconi, 2012) to object to the very narrow Coasian definition of PES (Wunder, 2005; Engel *et al.* , 2008). These authors argue that the majority of PES examples presented in the literature (e.g. Wunder *et al.* , 2008) only qualify as "PES-like", and there is no underlying reason why PES should aspire to a Coasian design. An alternative definition is presented by Muradian *et al.* (2010) as: "*a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources*". This alternative formulation accommodates a range of PES models, including direct public investment (government as buyer) or compensation schemes (polluter as funder), and emphasizes that the term PES implies *payments*, not *markets.*

Cross-border flows during the transformation and consumption of environmental goods and services complicates impact assessment and requires a methodology that is both flexible, accurate and globally applicable. Product-level Life Cycle Assessment (LCA) is one widely employed tool for assessing the potential environmental impacts of products and services from cradle to grave. LCA can deal with very complex production systems by accounting for flows of basic resources and energy, both directly and indirectly associated to the production process. However, spatial considerations and regionalization of impact assessment is currently limited in LCA, and the inclusion of biodiversity assessment is a very recent trend that is still in a fluid state of development across the multiple of impact categories in LCA.

1.4.2 How much compensation is enough?

Balanced and fair compensation using restoration or averted-loss offsets requires both sound conceptual theory, and a robust, yet flexible, methodology. Offsets represent a relatively new development in biodiversity conservation policy, both in the public and private domain. There have been prolific, widespread and valid criticisms of the theory, methodology and mode of implementation of offsets by both natural and social scientists in recent years (Moilanen *et al.* , 2009a; Maron *et al.* , 2012; Bekessy *et al.* , 2010; Bull *et al.* , 2012; Gibbons & Lindenmayer, 2007; Morris *et al.* , 2006; Walker *et al.* , 2009b; Robertson, 2004). This has illuminated substantial challenges for the design of even local, let alone global, offsetting schemes. However, many of these criticisms are aimed at constructively pinpointing (numerous) areas for improvement. To date, no agreed-upon methodology has been developed for calculating standardized ratios and multipliers, integrating spatial and temporal factors, dealing with uncertainty, identifying "no-go" areas, and aligning offsets with broader conservation priorities (e.g. Moilanen *et al.* , 2009a; Maron *et al.* , 2012; Overton *et al.* , 2013; Blundell & Burkey, 2007; Moilanen, 2012a). Resolving some of these issues would be required in order to propose even a rudimentary global biodiversity compensation mechanism.

1.4.3 There is no silver bullet for conservation

Efforts to halt and reverse biodiversity loss in the face of intense resource exploitation in the developing world are currently insufficient and underfunded. A range of new conservation strategies and mechanisms have been proposed in addition to public protected areas, but knowledge on their effectiveness, social fairness and cost-effectiveness is patchy at best (Ferraro, 2011; Miteva *et al.* , 2012; Ferraro & Pattanayak, 2006). Implementation of a compensation project in any part of the world where biodiversity conservation is urgently needed will require careful consideration of the socio-economic consequences associated with different strategies and implementation models. Adequate feasibility studies are thus required on a case-by-case basis, taking account both ecological optimization, cost-effectiveness, cultural norms and distributional fairness (Fischer *et al.* , 2009).

1.4.4 Opaque international supply chains and targeting under ambiguity

While global conservation priorities are relatively well researched (Brooks *et al.* , 2006b; Keith *et al.* , 2013), methods for balancing conservation trade-offs are not yet as detailed as national and subnational spatial conservation planning studies (Moilanen *et al.* , 2011a; Pouzols *et al.* , 2012). Large-scale (i.e. regional, continental, global) models of ecological value and cost effectiveness are only beginning to emmerge, limiting the analysis of cost-effectiveness in the use of conservation resources (Naidoo & Iwamura, 2007a; McCarthy *et al.* , 2012; Wuenscher & Engel, 2012). The origin of many traded commodities is often unknown, even at the national scale, and therefore targeting sub-national regions (e.g. WWF Ecoregions; Olson *et al.* , 2001) for receiving compensation is likely to be difficult, or require a probabilistic approach towards determining the location of production (e.g. Monfreda *et al.* , 2008). Where the location is unknown or ambiguous, a spatial targeting system is required to select the ecologically and economically most effective option based on available choices. Ideally, the level of detail of the (economic, ecological, socio-political) data in the tool, and complexity of analysis would scale to the amount of certainty of the origin of impact, thereby combining global and regional techniques.

1.5 Organisation of the dissertation

1.5.1 Research aims

The PhD thesis aims to develop a methodological framework for the implementation of an international compensation scheme for land use impacts on biodiversity. The dissertation will investigate the four issues identified above, particularly focusing on the latter three (impact assessment, conservation effectiveness and compensation methodology), and attempt to propose solutions to outstanding issues, or at a minimum discuss uncertainties, trade-offs and research needs. In doing so, the dissertation will contribute to a deeper understanding of the feasibility, ecological effectiveness, economic efficiency and development effects of an international biodiversity compensation scheme. The thesis is structured as five cumulative peer-reviewed publications, each forming a chapter (*Chapters 2–6*; Figure 1.2). A concluding chapter summarizes the main findings, and discusses the practical and academic relevance of the work in relation to similar ideas and approaches in the literature. The conclusion also highlights areas for improvement and further research. The proceeding chapters are thematically structured as follows:

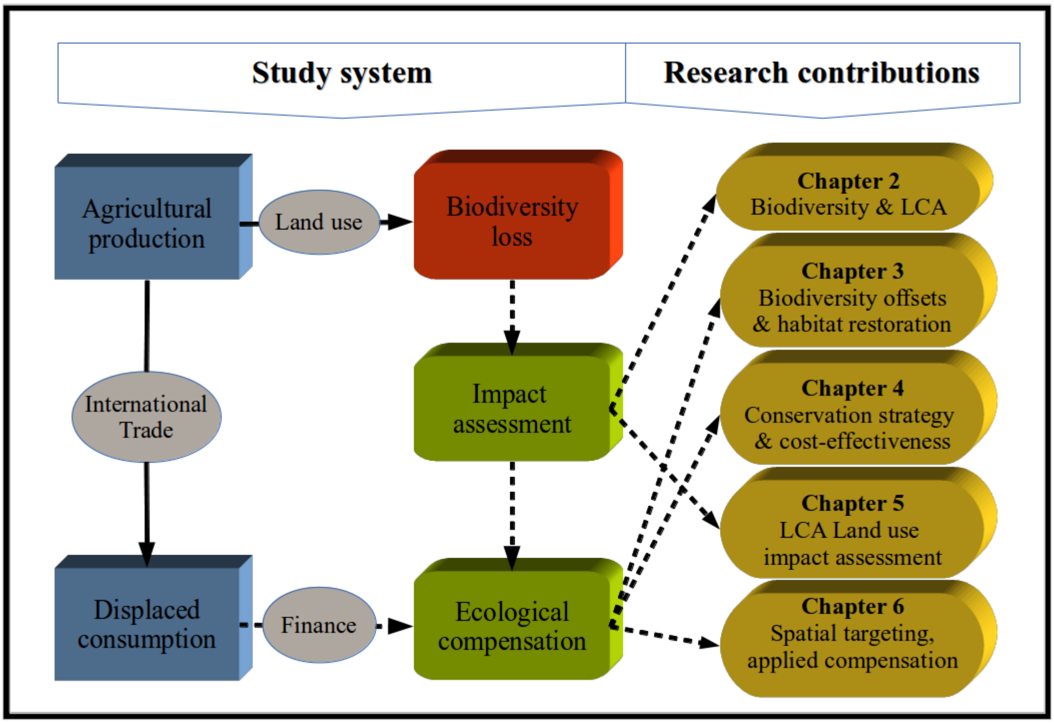

Figure 1.2 – **Structure of the dissertation**. Diagram illustrating the system under study and the specific research contributions (Chapters 2–6). The concluding chapter is not illustrated in the diagram.

1.5.2 Chapters and contributions

- **Chapter 2** focuses on the impact assessment stage of the framework: Quantifying the biodiversity impacts of products and services using Life Cycle Assessment (LCA). The chapter reviews available methods in Life Cycle Assessment across a wide selection of impact categories related to the five principal drivers of biodiversity loss identified by the Millennium Ecosystem Assessment (MA, 2005c): Terrestrial and aquatic habitat change, climate change, pollution, overexploitation and biotic homogenization. The chapter assesses how well LCA reflects these different drivers, and also how existing methods represent the different hierarchical levels (genes, species, communities, and ecosystems) and biological attributes (structure, function, and composition) encompassed by the term biological diversity. The chapter links these findings to a large and comprehensive review of concepts and methods in ecological research to help LCA practitioners develop new and meaningful indicators of damages to the biological diversity of the planet.

- **Chapter 3** shifts to the other side of the compensation framework, investigating the validity and empirical support for current offset policy. The chapter reviews over one hundred studies from the restoration ecology and habitat succession literature, extracting response and predictor data to help explain patterns in diversity over time following habitat disturbance. The chapter builds a robust statistical model of this empirical data to predict the recovery time of biodiversity across taxonomic groups and world regions, quantifying the benefit of active (i.e. human-assisted) restoration in comparison to a passive (natural succession) baseline. This model is used to derive estimates of restoration failure risk and time lags associated with the creation of biodiversity value through restoration offsets. The chapter then contrasts the research findings to current offset policy, in terms of developing robust offset ratios and offset multipliers that account for uncertainty, risk, and time lags, and comparing this to what is applied in practice.

- **Chapter 4** focuses on the implementation of conservation projects that would make up the compensation activities. The chapter specifically contributes to the literature on cost-effective conservation planning. Taking a case study region of Central Kenya (because of its relevance to export markets of agricultural commodities such as tea, coffee, tobacco, flowers etc.), the chapter investigates economic and ecological trade-offs between implementing two increasingly common strategies to protect biodiversity: Land Purchases and Easements (LPE) and Payments for Ecosystem Services (PES). The chapter develops a spatial economic-ecological model, and a site selection algorithm, to simulate spending a hypothetical conservation budget on securing sites for conservation while applying a cost-effectiveness criterion. The research plots how investment and operational costs accumulate over a 30 year time horizon and compares the performance and efficacy of the two strategies, with reference to potential development effects and institutional barriers to implementation.

- **Chapter 5** returns to the impact assessment issue, focusing specifically on land use impacts to biodiversity in LCA, and constitutes the only co-author chapter of dissertation. It builds on recommendations from *Chapter 2* regarding land use. The method is based on recently developed habitat suitability maps for all extant mammals, and develops an impact assessment method based on the weighted loss of species richness in Eastern Africa (the case study region). Richness decreases, weighted by species global rarity and threat status, are estimated for different types of agricultural land use, using a baseline of potential (historical) diversity patterns. The chapter specifically focuses on crops relevant for export to the developed world. Uncertainty in impact location for each crop is dealt with using a probabilistic approach employing production data from the Food and Agricultural

Organization (FAO). The method is compared to two previous impact assessment methods in LCA. For each method, occupation and transformation impacts are quantified per crop, and projected spatially over the study region. Resulting patterns in damages are interpreted with reference to the planning of agricultural production and expansion while meeting global targets of reducing the rate of global biodiversity loss.

- **Chapter 6** employs results and recommendations from the previous chapters in order to designs an international compensation framework for offsetting the biodiversity impacts of land use in East Africa. The framework is based on (i) the land use impact assessment method developed in *Chapter 5* (ii) the restoration offset model developed in *Chapter* 3, predicted spatially across the study region, and (iii) the cost models developed in *Chapter 4*. The framework applies a simplified interpretation of "robustly fair offset ratios" (Moilanen *et al.* , 2009a) to modelled biodiversity impacts using a biodiversity value function that represents the inverse of the biodiversity loss function from *Chapter 5* (i.e. loss and gain are measured using the same unit). A targeting step uses spatial conservation planning techniques to embed the compensation activities into regional (East Africa) and local (Central Kenya) conservation priorities. The study quantifies the predicted unit costs of compensating land (in \$/ha) and its derived economic product (\$/tonne agricultural commodity). The chapter concludes by estimating likely effects on both farm-gate and consumer prices of implementing a "biodiversity neutrality" policy to agricultural products.

- **Chapter 7** reflects on the findings of the entire dissertation, critically reviewing the practical and academic relevance of the research, and contrasting approaches and findings to similar research within the ecological economics and conservation literature. The chapter highlights weaknesses of the proposed methodology, future research directions to address these, and discusses potential options for application in the North–South context.

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Chapter 2

Toward meaningful endpoints of biodiversity in Life Cycle Assessment

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Abstract

Halting current rates of biodiversity loss will be a defining challenge of the 21st century. To assess the effectiveness of strategies to achieve this goal, indicators and tools are required that monitor the driving forces of biodiversity loss, the changing state of biodiversity, and evaluate the effectiveness of policy responses. Here, we review the use of indicators and approaches to model biodiversity loss in Life Cycle Assessment (LCA), a methodology used to evaluate the cradle-to-grave environmental impacts of products. We find serious conceptual shortcomings in the way models are constructed, with scale considerations largely absent. Further, there is a disproportionate focus on indicators that reflect changes in compositional aspects of biodiversity, mainly changes in species richness. Functional and structural attributes of biodiversity are largely neglected. Taxonomic and geographic coverage remains problematic, with the majority of models restricted to one or a few taxonomic groups and geographic regions. On a more general level, three of the five drivers of biodiversity loss as identified by the Millennium Ecosystem Assessment are represented in current impact categories (habitat change, climate change and pollution), while two are missing (invasive species and overexploitation). However, methods across all drivers can be greatly improved. We discuss these issues and make recommendations for future research to better reflect biodiversity loss in LCA.

2.1 Introduction

The planet is undergoing extensive changes induced by human appropriation of natural resources. Among the most critical consequences is the stark decline in biological diversity documented over the past decades (CBD, 1993; MA, 2005c; Alkemade *et al.* , 2009; Sala *et al.* , 2000). The Millennium Ecosystem Assessment (MA, 2005c) recently documented a widespread decline in the quality of global ecosystems and biodiversity, stimulating the rapid development of indicators to measure the changing state of nature, the driving pressures behind, and evaluate the effectiveness of policy responses. The MA identified a number of direct drivers of biodiversity loss, of which the most important are i) terrestrial and aquatic habitat change, ii) invasive species, iii) overexploitation of wild populations, iv) pollution, and v) climate change (MA, 2005c).

The development and use of assessment tools to track hot-spots of environmental damages in production systems has been of growing interest amongst industry, the public sector, and non-governmental organizations. One such method of environmental assessment at the product level is Life Cycle Assessment (LCA). LCA is used to quantify the potential environmental impacts throughout a product's life cycle from raw material acquisition, production, use, and finally disposal (Finnveden *et al.* , 2009). The "impact assessment" stage of LCA models impacts along mostly linear, deterministic, cause-effect chains by linking inventory items to so-called midpoint impact categories, such as global warming potential, ecotoxicity and land use. In an optional second step, the cause-effect chain is extended to final endpoints, which express impacts on three areas of protection: natural resources, human health and ecosystem quality.

The development and inclusion of potential endpoints for biodiversity in LCA has been ongoing for more than a decade (Goedkoop & Spriensma, 2001; Lindeijer, 2000a; Weidema & Lindeijer, 2001; Blonk *et al.* , 1997; Heijungs *et al.* , 1997). Yet many methods in LCA are still in early stages of development (Koellner & Scholz, 2008; Schmidt, 2008; Pfister *et al.* , 2009; De Schryver *et al.* , 2009; van Zelm *et al.* , 2007a; Larsen & Hauschild, 2007; Pennington *et al.* , 2004a; Pelletier *et al.* , 2007; Koellner & Scholz, 2007). Particular difficulties are posed by the methodological framework of LCA itself, which traditionally required impacts to be generic in space, summed across time horizons, strongly linked to a functional unit, and free of interactions between impact pathways (Udo de Haes, 2006). These restrictions are only beginning to be addressed by recent LCA research. If LCA is to be truly informative to decision makers about biodiversity loss, it is important to assess whether current methods are reflecting i) the major drivers of biodiversity loss as identified by the MA and ii) whether they capture the concept of biodiversity adequately both in its inherent variation, and its non-uniform distribution across the planet.

This review article synthesizes how biodiversity indicators are currently employed in LCA during endpoint cause-effect modelling, and how this could be improved in the future by drawing on available methods in biodiversity research. We begin by describing a framework for characterizing biodiversity indicators. We then assess the coverage of biodiversity in relevant impact categories of LCA in light of this framework. We highlight research gaps and offer suggestions for improvement based on a review of a wide range of methods employed in the biodiversity assessment literature, including a comparison of LCA with recent biodiversity assessments such as the MA (see Appendix 8.1.7).

2.2 Biodiversity indicators

Framework. We adopt the definition of biodiversity provided in the Convention on Biological Diversity (CBD, 1993), as a nested hierarchy of components defined by the level of scale and complexity (Noss, 1990). We separate components into four broad levels: gene, species, community and ecosystem (Noss (1990); Niemi & McDonald (2004); differing slightly from the CBD). At each level, components may be characterized in terms of attributes, which reflect composition, the quantity and variety of elements; function, the ecological and evolutionary process acting amongst elements; and structure, the physical organization of elements (Niemi & McDonald, 2004). Indicators used to measure biodiversity can be correspondingly described by attribute and component. Table 2.1 illustrates the indicator framework adopted for this review.

Genetic indicators. At the genetic level, indicators reflect intraspecific (within-species) or interspecific (between species) diversity (Appendix 8.1.1). Intraspecific indicators include heterozygosity, allelic richness, nucleotide diversity, genetic variance and heritability (Hughes *et al.* , 2008). Interspecific variation is quantified using phylogenetic indicators (Faith, 2002), including phylogenetic diversity (Faith *et al.* , 2004b).

Species-based indicators. Species level indicators describe trends in the abundance or attributes of individual species (Appendix 8.1.2). These include the focal species approach (Lambeck, 1997) and species intactness indices (Scholes & Biggs, 2005). Methods in modelling species distributions, habitat suitability and sensitivity to environmental stress may take a deductive approach, using expert opinion and meta-analysis to derive cause-effect relationships, or an inductive approach, extracting patterns from empirical data via multivariate statistics with no proposed a priori cause-effect mechanism (Corsi *et al.* , 2000).

Community indices. Community level indices describe the emergent patterns in biodiversity resulting from the overlap of individual species ranges (Appendix 8.1.3). They characterize the number and relative abundance of species in a community in a single value. The value of a diversity index may be totally dominated by the number of species in a community (i.e. species richness), or conversely by only the relative abundance of species (i.e. pure evenness indicators). Intermediate points represent well-known indices such as the Shannon-Wiener index and Simpson's concentration (Jost, 2006). Community indices such as the Bray Curtis index or ordination measures also represent beta diversity changes (turnover) between samples or locations. Recent work has focused on modelling emergent community patterns in biodiversity such as richness, turnover and endemism as a substitute for data-demanding species level approaches (Ferrier & Guisan, 2006) (Appendix 8.1.4). Indices of functional diversity may be derived by grouping species into functional groups, or calculating continuous distance in functional trait space (Petchey & Gaston, 2006).

Ecosystem and landscape indicators. Indicators of ecosystem diversity are split into those of pattern and process (McGarigal *et al.* , 2009) (Appendix 8.1.5). Landscape pattern indicators represent human-perceived patterns in a landscape (diversity, patch size, and configuration of habitat). Their link with biological processes, such as dispersal and persistence of species, is not fully understood (Turner, 2005). However, key pattern metrics such as area of native habitat and summed anthropogenic edge length often correlate with processes and patterns in emergent biodiversity (Fischer & Lindenmayer, 2007). Variables derived from remote sensing, such as the Normalized Difference Vegetation Index (NDVI), can illustrate disturbance regimes, measure vegetation cover, and chart phenological changes (Foody, 2008).

2.3 Assessing biodiversity loss in LCA

In the LCA framework, endpoint impacts on biodiversity resulting from an environmental intervention (e.g. emission of acidifying substances, conversion or occupation of land) have been developed in the impact categories of land use, water use, climate change, acidification and eutrophication, and ecotoxicity. This covers three of the five principal drivers of biodiversity loss as identified by the MA (Table 2.2). Endpoint models generally include three components: **i) fate factor**, which models the spatial distribution and intensity of pressures induced by an intervention; **ii) impact factor** (also known as damage or effect factor), which relates the intensity of a unit of pressure to a quantified loss in biodiversity, frequently expressed as the unit-less indicator of "*potentially disappeared fraction* (PDF) of species" (Goedkoop & Spriensma, 2001); and **iii) characterization factor**, which multiplies impact factors (intensity of the intervention) by fate factor (size and duration of the intervention) to give an endpoint of PDF*m^{x}*y, where the power term, x, equals 2 (for area fate models) and 3 (for volume fate models). Below we briefly summarize the history and state of art of endpoint modelling in each impact category.

2.3.1 Land use

The conversion of natural habitat to human use has been the most important driver of biodiversity loss over the past century (MA, 2005c). Methods to characterize land use impacts to biodiversity in LCA have mainly used compositional indicators at the level of the local community, primarily species richness (Goedkoop & Spriensma, 2001; Lindeijer, 2000a; Koellner & Scholz, 2008; Schmidt, 2008; Koellner, 2000, 2003; Geyer *et al.* , 2010b). Koellner and Scholz (2008) additionally used the number of threatened species as an indirect indicator of ecosystem diversity and land use value. Some studies have included ecosystem level indicators directly, using the relative area of ecosystems (Pelletier *et al.* , 2007; Vogtländer *et al.* , 2004; Kyläkorpi *et al.* , 2005; Müller-Wenk, 1998), or by predicting the effect of ecosystem level changes on the regional pool of species (Schmidt, 2008; Koellner, 2000). Genetic indicators are absent, but Weidema and Lindeijer (8) proposed calculating the change in community phylogenetic diversity due to land use change.

A first attempt to incorporate the effects of fragmentation at the midpoint level was made by Jordaan *et al.* (2009). Schenck (2001) presented a range of indicators

Hierarchical components	Biological attributes			Assessment tools and methods
	Composition	**Structure**	**Function**	
Genes (biotic)	heterozygosity, allelic diversity, % polymorphic loci, genetic variance, phylogenetic diversity	chromosomal or phenotypic polymorphisms, physical genetic distance, effective population size, generation overlap, heritability	mutational diversity, mutation rate, duplication rate, selection intensity, rate of genetic drift, gene flow	visible polymorphisms, molecular markers (protein electrophoresis), DNA markers (microsatellites, DNA sequencing), parent-offspring regression, sibling analysis
Species (biotic)	(meta)population size and number, absolute or relative abundance, frequency, biomass, cover, intactness, density	size, morphological variability, physiognomy, population structure, home range size and distribution in space, dispersal patterns, habitat requirements	metapopulation dynamics (drift, bottle necks, inbreeding, outbreeding trends) demographic processes (growth, reproductive, feeding, nesting, dispersal rate)	population censuses, time series analysis, remote sensing and GIS, habitat suitability index, species-habitat modelling, population viability analysis, species distribution modelling
Communities (biotic, abiotic)	species richness, relative abundance, higher taxon diversity, phylogenetic diversity, number of endemics, invasive, threatened or focal species, similarity and turnover of species assemblages	habitat structural complexity, foliage physiognomy and layering, habitat density, gap density, volume, surface area, slope, aspect, rugosity index, nearest neighbour distance	nutrient turnover and cascades, functional group or guild diversity, number and strength of interspecific interactions (predator-prey, parasite-host), biomass production, extinction, colonization rates	remote sensing and GIS, aerial photographs, time series analysis, physical habitat measures, observation habitat descriptions, multi-species, local sampling techniques, multivariate integrative indices (Shannon-Wiener index, dispersion, layering, biotic integrity)
Ecosystems and landscapes (abiotic)	patch diversity, richness, composition, number of ecosystems, relative or absolute area, area of semi-natural vegetation in agriculture, emergent patterns in species distribution (richness, endemism)	patch shape and configuration (fragmentation, isolation, connectivity, spatial linkage), patch size frequency distribution, topography, river and shoreline profile	disturbance patterns and regimes (frequency, extent, intensity, seasonality), pattern metrics (patch turnover), erosion potential, geomorphic and hydrologic processes, land use patterns and trends	remote sensing and GIS, aerial photographs, time series analysis, spatial statistics, mathematical indices (pattern, connectivity, heterogeneity, layering, edge extent, diversity, fractal measures and autocorrelation)

Table 2.1 – Indicator framework. Biodiversity indicators across hierarchical components (gene, species, community and ecosystem) and biological attributes (compositional, structural and functional). The final column on the right illustrates possible assessment methods (adapted from Noss, 1990; Niemi & McDonald, 2004).

including the protection of habitats/species, connectivity of habitats, invasive species, and percentage of native-dominated vegetation, but proposed no means to integrate these into an operational framework. Michelson (2008) included a limited set of local indicators of ecosystem function. Net primary productivity (NPP) was proposed early on as a functional indicator (Lindeijer, 2000a; Weidema & Lindeijer, 2001; Blonk *et al.* , 1997). Other approaches used the concept of hemeroby – naturalness of ecosystems – to characterize land use types on a scale of 0, purely artificial, to 1, no human influence (Baitz *et al.* , 2000; Bauer & Zapp, 2004; Brentrup *et al.* , 2002).

Vascular plant species richness has been the core taxa for land use assessments, mainly because of data availability and the close associations to specific land uses (Goedkoop & Spriensma, 2001; Koellner & Scholz, 2008; Schmidt, 2008; Koellner, 2000). Michelsen (2008) suggested an integrative index, Conditions for Maintained Biodiversity, in order to address taxonomic coverage, but this indicator requires region-specific information, not readily available for many areas or ecosystems. Koellner and Scholz (2008) included species richness of mollusks and moss in addition to vascular plants. Geyer *et al.* (2010b) used the habitat affinities of vertebrate species to calculate impacts on species richness, abundance, and evenness. Mattsson *et al.* (2000) recommended using richness of mammals, birds and butterflies, but no overall assessment framework was provided. A multi-taxa approach to agricultural land use and management regimes using taxon-specific impact scoring was employed in the Swiss Agricultural Life Cycle Assessment (SALCA) method (Jeanneret *et al.* , 2008).

The majority of approaches were developed for specific geographic regions, namely Northern Europe (Koellner & Scholz, 2008; Koellner, 2000; Michelsen, 2008), North America (Toffoletto *et al.* , 2007; Geyer *et al.* , 2010b) and South-East Asia (Schmidt, 2008). Weidema and Lindeijer (Weidema & Lindeijer, 2001) proposed a global approach at a biome scale, but it remains of limited application because of the coarseness of the method (but see Toffoletto *et al.* , 2007 for a regionalized adaptation). The species-area relationship (SAR) (Rosenzweig, 1995) and sample-based rarefaction (Colwell *et al.* , 2004a) have been applied to compare the species richness of standardized sampling areas in different land classes (Koellner & Scholz, 2008; Schmidt, 2008; Koellner, 2000). The SAR also forms the basis for regional damage calculation, estimating the increased risk of regional extinction due to land use change (Koellner & Scholz, 2008; Schmidt, 2008; Koellner, 2000).

2.3.2 Water use

Anthropogenic water use reduces regional availability, impairing the functioning and diversity of water-dependent terrestrial (Pfister *et al.* , 2009) and freshwater (Xenopoulos *et al.* , 2005) ecosystems. The assessment of water use impacts in LCA is a relatively new development. Mila i Canals *et al.* (2009) suggested an indicator, Freshwater Ecosystem Impact, to describe the damage to natural ecosystems resulting from depleted water resources via land occupation and the resulting changes in runoff quality and quantity, and abstractive use for agriculture. Pfister *et al.* (2009) developed the only existing method to model the impacts of freshwater use on natural ecosystems to the endpoint level. In this work, impacts to terrestrial biodiversity were approximated using water-limited NPP as a functional indicator (based on Nemani *et al.* , 2003). Links to species richness were established via a correlation between NPP and vascular plant species richness at low to medium levels of species richness (Pfister *et al.* , 2009). It was assumed that water-limited terrestrial ecosystems, generally in arid areas, do not display high species richness. Since water availability and vulnerability of ecosystems varies as a function of space, the method of Pfister *et al.* (2009) was regionalized to provide impact factors for all global watersheds.

2.3.3 Climate change

Climate change, driven by anthropogenic greenhouse gas emissions, is expected to cause a large number terrestrial extinction over the next century due to changing temperature, precipitation and seasonality (Thomas *et al.* , 2004; Thuiller *et al.* , 2006). Aquatic effects include extinctions of fish species due to reduced river discharge (Xenopoulos *et al.* , 2005) and mass extinctions of coral reefs due to warming sea temperature (Hughes *et al.* , 2003) and increased ocean acidification (CBD, 2009). Within LCA, the only operational impact assessment method for climate change was restricted to terrestrial biodiversity and is based on the increased extinction risk associated with changes in individual species' distributions under future climate scenarios (De Schryver *et al.* , 2009). The damage factor was based on the work of Thomas *et al.* (2004), a review of nine studies linking regional extinction risk with changing climate across a number of species groups.

The method of De Schryver *et al.* (2009) included assessments of 1084 plant and animal species across five regions (in Europe, Mexico, Australia, South Africa, and Brazil). Climate envelope modelling was used to estimate range area reductions per species and the associated increase in extinction risk (Thomas *et al.* , 2004). Three approaches based on the SAR were used to estimate predicted extinctions. Responses were tested under assumptions of dispersal and non-dispersal ability, and red list species and all species respectively (Thomas *et al.* , 2004). This species level approach was used to approximate the potentially disappeared fraction of species, extrapolated to global terrestrial extent to represent a global impact factor.

Acidification and eutrophication. Acidification and eutrophication leads to a disruption of the natural nutrient balance, altering the species composition of ecosystems, and frequently leading to a loss of biodiversity (Smith *et al.* , 1999; Stevens *et al.* , 2004). The effects of acidification on ecosystems quality have been included in LCA using methods that considered the sensitivity of the receiving ecosystems (Hauschild & Potting, 2005; Posch *et al.* , 2008; Krewitt *et al.* , 2001; Seppälä *et al.* , 2006; Potting *et al.* , 1998; Huijbregts *et al.* , 2000b), or effects on NPP (Hayashi *et al.* , 2004a). Species level impacts for terrestrial acidification have been modelled based on occurrence data for vascular plants (Goedkoop & Spriensma, 2001; van Zelm *et al.* , 2007a; Goedkoop *et al.* , 2009) and butterflies (Goedkoop & Spriensma, 2001). Van Zelm *et al.* (2007a) modelled the probability of occurrence of over 240 forest plant species in Europe. A threshold was constructed for each species as a function of the base saturation of the soil. The PDF was approximated by the percentage of species predicted to be absent because of elevated base saturation. This impact factor was multiplied by the total forest and non-forest area of Europe to approximate overall impacts to the region affected by acidifying emissions.

Freshwater eutrophication was included in the ReCiPe method (Goedkoop *et al.* , 2009) for Europe using the diversity of macro-invertebrate genera as indicators of taxonomic diversity. Environmental modelling was used to predict the absence of over 837 macro-invertebrate genera, approximating PDF, as a function of phosphorus concentration (Goedkoop *et al.* , 2009).

Taxonomic coverage for both acidification and eutrophication remains limited to species groups where pressure and occurrence data exist. Current methods are applicable to Europe, although the impact factors are likely to apply to other temperate climates, but not to tropical and sub-tropical regions (Bobbink *et al.* , 2010). Impact factors for aquatic acidification and marine eutrophication are absent from any proposed method.

2.3.4 Ecotoxicity

Chemical emissions to aboveground biomass, air, water, and soil cause toxicity to a variety of organisms. Research on ecotoxicological impacts to biodiversity in the context of LCA has been ongoing for many years (Larsen & Hauschild, 2007). Model species in laboratory settings are used to establish the potentially affected fraction (PAF) of species due to elevated concentrations of a toxin. It is defined as the percentage of species within a community or taxonomic group that is expected to be exposed above a certain effect-related threshold, such as the effect concentration for 50 percent of the population (EC50) or the no observed effect concentration (NOEC) (Larsen & Hauschild, 2007; de Zwart & Posthuma, 2005). This uses species level indicators of abundance and reproductive decline.

The exact relationship between the PAF and species loss from a community is a topic of debate. By comparing laboratory experiments with field data, Van den Brink *et al.* (2002) showed that direct effects of long-term and acute exposure are generally well reflected by species sensitivity distributions used to calculate the PAF of a community. The study assessed changes in biodiversity across a wide range of animal and plant taxa. Posthuma and de Zwart (2006) showed that in fish species assemblages in North American streams, the observed loss of species ascribed to mixture toxicity closely matched the predicted risks based on EC50. Snell and Serra Snell & Serra (2000) modelled reproduction effects on rotifers, and showed that an EC50 will result in population extinction after a long exposure time. The PAF therefore may have the diagnostic properties required to assess ecological responses to ecotoxic stress.

LCA models are available that can be adapted to meet region-specific conditions, but can also provide continental and global factors, such as the USEtox model (Rosenbaum *et al.* , 2008). Freshwater biodiversity responses have received the most attention, and more research is necessary on the response of terrestrial and marine ecosystems. Taxonomic coverage is usually limited to low trophic position, cold-blooded species. The effects of bioaccumulation and biomagnification are only beginning to be investigated in LCA.

2.3.5 Limitations

Endpoint modelling in LCA currently suffers from at least two classes of limitation, that we term “Conceptual limitations” and “Data limitations”. The former relates to methodological choices during impact factor development and the constraints imposed by the overarching LCA framework. The latter relates to a general lack of knowledge on how interventions affect biodiversity, and a lack of biological data for many taxa and world regions. This has invariably necessitated the adoption of numerous assumptions, some of which are listed below, split into conceptual limitations (1–4) and data limitations (5–7):

1. **Endpoint unit**. Because PDF is unit-less, characterization factors (PDF*m^{x}*yr) are expressed in units of area/volume loss (in m^2 or m^3) for a fixed duration (in yrs). This assumes impacts across all impact categories can be expressed as an effective loss of habitat for biodiversity.

2. **Impact scale.** Some impact factors express local extinctions (e.g. acidification) while others express regional extinctions (e.g. climate change). It is assumed that impacts at different scales can be directly compared and aggregated.

3. **Linear damage relationship.** When characterization factors are developed for scales other than what was used for impact factor development, the relationship

between species loss (overall impact) and area affected is assumed to be linear (except for the land use regional impact which applied the non-linear SAR; Koellner & Scholz, 2008).

4. **Use of indicators.** Indicators used to construct impact factors may reflect various components (species, communities, and ecosystems) and attributes (compositional, functional, and structural) of biodiversity, but are assumed to approximate damages to species diversity (PDF).

5. **Geographic coverage.** Regionalization is incomplete for most impact factors. It is assumed that impact factors developed in certain regions (mainly Western Europe) are representative of all world regions. Furthermore, characterization factors are missing for major realms across all impact categories (e.g. marine eutrophication, terrestrial ecotoxicity, aquatic lake- and sea-bed damage).

6. **Taxonomic coverage.** Only a relatively small number of taxa are used to develop impact factors. These impact factors are assumed to apply to all taxa during the construction of characterization factors.

7. **Missing drivers.** Two drivers of biodiversity loss (overexploitation and invasive species) are completely missing from LCA. Until impact pathways for these drivers are developed, their importance regarding biodiversity loss is essentially assumed to be negligible.

In the following sections, we critically examine these assumptions and highlight potential research directions that could improve biodiversity assessment in LCA. Due to space limitations, the last assumption (missing drivers) is addressed in the Appendix (section 8.1.8).

2.4 Endpoint unit, scale, and linearity

2.4.1 Endpoint unit

Endpoints of biodiversity in LCA are expressed across all impact categories as an effective loss of habitat, which converts the area/volume partially affected by an intervention (i.e. PDF < 1) to an equivalent area/volume of total loss of habitat value for biodiversity (PDF $= 1$). The exact proportion is dependent on the intensity of the intervention, which is given by the impact factor. This is a potentially useful approach that has been applied outside of LCA to assess land use impacts using variants of the SAR (Scholes & Biggs, 2005; Pereira & Daily, 2006) and forms the basis of the GLOBIO3 modelling framework

MA driver and corresponding LCA impact category	**Modelling approach (data type)**	**Indicator**		**Taxonomic coverage**	**Geographic coverage**
		Component	**Attribute**		
HABITAT CHANGE					
Land use					
Koellner (2000; 2003); Koellner and Scholz (2008); Schmidt (2008)	standardized species richness; standardized number of threatened species (EDP) (sampled species occurrence data)	community	composition	vascular plants, mollusks (EDP), moss (EDP)	C. Europe, SE Asia (Schmidt Schmidt (2008))
Koellner (2000; 2003); Schmidt (2008)	SAR-based; proportion of LI land in region and assumed species associations (GIS vegetation/LU classes)	ecosystem	composition		
Michelsen (2008) (*cf.* Weidema and Lindeijer, 2001)	area index for scarcity; integrative conservation index for vulnerability (WWF Ecoregion data)	ecosystem	integrative	weight for IUCN threatened species	global
Michelsen (2008)	Conditions for Maintain Biodiversity index; decaying wood (regional estimates), set aside land (area), invasive species (percentage community)	ecosystem	integrative	n/a (multiple proxies used)	Scandinavian and Russian taiga
Water use					
Pfister *et al.* (2009)	environmental modelling; w-NPP change due to water use (remote sensing and GIS)	ecosystem	function	n/a (w-NPP proxy for vascular plant richness)	global

Endpoint modelling (second column) refers to method and data used to calculate characterization factors. Indicator component and attribute is described in the text. Taxonomic coverage lists species groups assessed. Geographic coverage refers to broad region used in analyis. PDF = potentially disappeared fraction of species; PAF = potentially affected fraction of species; SSD = species sensitivity distribution; HC50 = effect concentration for 50% of the population; NOEC = no observed effect concentration; BS = base saturation; SAR = species area relationship; w-NPP = water limited net primary productivity; EDP = Ecosystem Damage Potential, LI = low intensity; n/a = not applicable.

Table 2.2 – LCA biodiversity models. Recent methods in LCA that model endpoints of biodiversity loss cover three of the five drivers identified by the MA. Indicators generally reflect compositional aspects of biodiversity, and are limited both taxonomically and geographically.

MA driver and corresponding LCA impact category	**Modelling approach (data type)**	**Indicator**		**Taxonomic coverage**	**Geographic coverage**
		Component	**Attribute**		
CLIMATE CHANGE					
De Schryver *et al.* (2009)	meta-study-based impact factor; climate-envelope modelling; 3 SAR-inspired methods (species occurrence data)	species	composition	vascular plants, mammals, birds, amphibians, insects	global (extrapolated)
POLLUTION					
Acidification and eutrophication"					
Van Zelm *et al.* (2007a) (acidification)	modelled species absence as function of BS (species occurrence data); % sp. absent $\approx$ PDF	species	composition	vascular plants (forest species)	Europe
Goedkoop and Spriensma (2001) (acidification and eutrophication)	modelled species absence as function of nitrogen deposition (species occurrence data); % sp. absent $\approx$ PDF	species	composition	vascular plants, insects (butterflies)	Netherlands
Goedkoop *et al.* (2009) (eutrophication)	modelled genera absence as function of phosphorus conc. (genera occurrence data): % gen. absent $\approx$ PDF	species	composition	insects (macro-invertebrate)	N. Europe
Ecotoxicity					
Van den Brink *et al.* (2002); Posthuma and de Zwart (2006)	testing the PAF $\approx$ PDF relationship with semi-field data (community composition and richness)	community	composition	cold-blooded model organisms	temperate climates
Goedkoop and Spriensma (2001)	NOEC(SSD)-based PAF $\approx$ 0.1*PDF; laboratory species reproductive declines (species abundance data)	species	composition	cold-blooded model organisms	
Rosenbaum *et al.* (2008)	USEtox; HC50(SSD)-based PAF; laboratory species reproductive declines (species abundance data)	species	composition		global

Endpoint modelling (second column) refers to method and data used to calculate characterization factors. Indicator component and attribute is described in the text. Taxonomic coverage lists species groups assessed. Geographic coverage refers to broad region used in analyis. PDF = potentially disappeared fraction of species; PAF = potentially affected fraction of species; SSD = species sensitivity distribution; HC50 = effect concentration for 50% of the population; NOEC = no observed effect concentration; BS = base saturation; SAR = species area relationship; w-NPP = water limited net primary productivity; EDP = Ecosystem Damage Potential, LI = low intensity; n/a = not applicable.

Table 2.2 cont.

(Alkemade *et al.* , 2009). It has also been expanded to include freshwater impacts across a range of pressures using river section length in place of area (Turak *et al.* , 2011).

2.4.2 Impact scale

Current LCA impact factors estimate species extinctions, in terms of PDF, at largely undefined spatial scales. Biodiversity loss represents a concern over the potential extinction of species at broadly defined scales (sub-national, national, and international; CBD, 1993). Likewise, the scale of extinctions in LCA requires standardization at one or multiple scales (i.e. the local community, the ecosystem or landscape, the region, or the globe). Koellner (2003) highlighted the arguable distinction between local impacts, which reflect concerns over the loss of local ecosystem functioning, and regional impacts, which reflect conservation concerns over species loss. Both are valid impacts, but convey very different messages to the end user of an LCA. This raises serious questions about the validity of current aggregative single score assessment tools where damages across impact categories, representing various geographic scales of extinction, are combined by simple summation (e.g. EcoIndicator 99 and ReCiPe 2008).

2.4.3 Linear damage relationship

Current scaling of impact factors to arrive at characterization factors assumes a simple linear damage function (PDF*area). This should be reconsidered given the importance of scale influences in ecology (Wiens, 1989) and the presence of non-linearity's, tipping points, and critical thresholds in biological responses to disturbance (Swift & Hannon, 2010). The adoption of the non-linear SAR and its variants (Pereira & Daily, 2006; Faith *et al.* , 2008; Koh & Ghazoul, 2010a; Turak *et al.* , 2011) could instead be used to take advantage of assumption one (endpoint unit) in order to address assumption two (impact scale) and three (linear damage relationship). This would require an extra step in characterization factor development that would consist of expressing effective habitat loss as a reduction in the species pool of the affected ecosystem(s), thereby providing the fraction of species potentially lost at the defined scale. This percentage loss could be related to absolute species losses using widespread regional checklist data for various taxa (e.g. WWF Ecoregions; Olson *et al.* , 2001).

2.5 Use of indicators

Indicators in LCA tend to reflect compositional changes in biodiversity, particularly at the species and community level (Table 2.2). Indicators that reflect changes in a variety of components (species, communities, ecosystems) and attributes (composition, structure, function) of biodiversity are also often employed to approximate species loss in terms of PDF. For example, the SAR is used to translate ecosystem indicators of habitat area change into predicted species losses. Likewise, single-species indicators are combined to approximate overall impacts across all species, either in a local community (e.g. PAF in ecotoxic impacts) or across an entire region (e.g. climate change impacts). Additionally, the functional indicator of NPP change is used as a proxy for species loss in water use impact assessment. Below we discuss the implications of this approach, and the possibility of developing new impact factors to reflect additional aspects of biodiversity.

2.5.1 Genetic component

Genetic indicators are absent from LCA. Developing impact factors based on phylogenetic diversity would enable interspecific genetic diversity to be approximated using existing species data (Faith, 2002; Faith *et al.* , 2004b). Cadotte *et al.* (2008) demonstrates an approach to calculate changes in phylogenetic diversity for plant communities using molecular sequence data from GeneBank (www.cnbe.nlm.nih.gov). The impact factor would express changes in the sum of branch lengths linking species from a sample of a community (Faith *et al.* , 2004b). Intraspecific (within-species) genetic variation, reflecting impacts to the genetic diversity of single species, such as population declines leading to reduced heterogeneity, will be extremely difficult to incorporate (see Appendix 8.1.1), and is limited by data availability (Laikre, 2010).

2.5.2 Species and community component

By definition, PDF is a multi-species index, and aggregating single-species indicators assumes that all species react to pressures in the same way as those assessed (Appendix 8.1.4). As the number of assessed species increases, results should be expected to converge (Roberge & Angelstam, 2004; MA, 2005a). In species level approaches in LCA, the number of species employed to construct impact factors is generally large (e.g. 1084 species for climate change (De Schryver *et al.* , 2009); 240 species for acidification (van Zelm *et al.* , 2007a); 837 genera for eutrophication (Goedkoop *et al.* , 2009)). In ecotoxicity the use of model organisms generally does not exceed ten species per substance (van Zelm *et al.* , 2009). This introduces large uncertainties and more studies are needed that test these laboratory results against field data (e.g. (Posthuma & de Zwart, 2006; Snell & Serra, 2000)). Inclusion of both species and community approaches across impact categories may offer better estimates of trade-offs and uncertainties associated with different methods.

2.5.3 Ecosystem and landscape component

In LCA, ecosystem indicators used to approximate species loss at the endpoint have been employed only in land use, and consider relatively simple effects based on the SAR (Koellner & Scholz, 2008; Schmidt, 2008; Koellner, 2000, 2003). Recent land use methods are including more complex models of habitat area and composition (Geyer *et al.* , 2010b). Outside of LCA, the InVEST tool (Nelson *et al.* , 2009) takes a detailed approach in estimating the contribution of each habitat patch to a species' persistence in the landscape using "countryside SARs" (Pereira & Daily, 2006). This accounts for patch size, cumulative anthropogenic habitat edge length, configuration, and the habitat requirements and dispersal ability of the assessed species (Nelson *et al.* , 2009).

The current species-level approach toward climate change modelling in LCA could be expanded to employ ecosystem-level indicators by modelling ecosystem area changes and resulting species loss via the SAR. For example, the MA (2005a) and GLOBIO3 (Alkemade *et al.* , 2009) predict biome and vegetation community expansions and contractions under IPCC scenarios using the IMAGE model (MNP, 2006), and relates this to species loss predicted by the SAR (Alkemade *et al.* , 2009; Hooper *et al.* , 2005). The ecosystem impact of consumptive water use is modelled to impacts on vascular plant species diversity in LCA through water-limited NPP. Additional, direct impacts of water use on fish species richness of rivers could be captured by the species-discharge relationship (Oberdorff *et al.* , 1995) both for water use and climate change (Xenopoulos *et al.* , 2005). Remote sensing data could help expand this to identify terrestrial drought damages to ecosystems caused by water abstraction or climate change (Foody, 2008).

Ecosystem effects of acidification, eutrophication, and ecotoxicity, modelled to the endpoint of species loss, are lacking. Ecosystem impacts have been developed in LCA using critical nitrogen and phosphorous loading values for acidification and eutrophication (Posch *et al.* , 2008; Seppälä *et al.* , 2006), but the consequential effects on species richness were not included. Critical loading has been extensively employed outside of LCA (e.g. Smith *et al.* , 1999; Bobbink *et al.* , 2010). Acidification and eutrophication are covered in both the MA (2005a) and GLOBIO3 model (Alkemade *et al.* , 2009) using meta-analyses of empirical studies documenting the relationship between exceedance of critical load and species loss (e.g. Bobbink, 2004). Such a relationship could be adapted to existing LCA methods.

2.5.4 Biological attributes

The majority of indicators of biodiversity in LCA measure composition (Table 2.2). Indicators of structure and function are largely absent. There are a range of indicators and methods to infer structural information at the local to ecosystem scale (Table 2.1). Impacts such as eutrophication or land use cause extensive structural alterations to habitats. A meta-analysis of published studies documenting the effect of pressures on the structural diversity of communities could potentially yield impact factors which could be used as a rough proxy for species loss (Gardner *et al.* , 2010).

On an ecosystem level, indicators were used in LCA to reflect fragmentation effects at the midpoint of land use (Jordaan *et al.* , 2009), but no attempt was made to model the damage to species richness. The meta-study of Harper *et al.* (2005) could be used to complete this process. Outside of LCA, fragmentation effects were included in the GLOBIO3 model using 6 published datasets that quantify species loss as a function of patch size (Alkemade *et al.* , 2009). The BioScore tool (Louette *et al.* , 2010) uses focal species that are sensitive to fragmentation (e.g. habitat specialists).

Changes in functional diversity are currently considered in LCA only at the ecosystem level using NPP. Abiotic indicators, such as NDVI-based metrics, hydromorphic and geomorphic modelling, erosion potential, and disturbance indicators could further be used to model impacts to ecosystem functioning (functional diversity; Table 2.1). The relationship between species diversity and ecosystem function is not well enough understood to allow PDF to act as a proxy for functional diversity, or *vice-versa* (Hooper *et al.* , 2005). New methods in land use developed in the framework of the UNEP-SETAC Life Cycle Initiative will employ abiotic functional indicators to model damages to a separate functional endpoint for ecosystems services (Wittstock *et al.* , 2008). Thuiller *et al.* (2006) modelled climate change impacts to functional group diversity of plants at the community level. This could be incorporated into LCA directly as a new impact factor for climate change. Interestingly, Cadotte *et al.* (2008) found phylogenetic diversity to be a better predictor of ecosystem function than functional group diversity. This might indicate how functional and genetic diversity may be incorporated into LCA using a single indicator.

2.5.5 Multiple impact factors

In order to better reflect the diverse components and attributes of biodiversity, we see the need to develop multiple impact factors for biodiversity. Currently in LCA, compositional indicators at the species, community, and ecosystem level approximate PDF. Structural indicators at the community and ecosystem level may also be expressed in terms of PDF, such as reductions in habitat complexity, increased fragmentation and habitat patch configuration (Alkemade *et al.* , 2009; Nelson *et al.* , 2009). Genetic diversity will require a separate impact factor of phylogenetic diversity. Phylogenetic diversity may also function as a good proxy for community functional diversity; otherwise functional groups or trait-space distance could be used to create an additional impact factor. Finally, community and ecosystem functional diversity will require an independent impact factor (e.g. damages to ecosystem services; Zhang *et al.* , 2010a). Including the genetic component of function and structure, and the species component of structure into LCA is not foreseeable in the near future.

2.6 Taxonomic and geographical coverage

2.6.1 Taxonomic coverage

Methodologies for all impact categories (except climate change) were developed using very few taxonomic groups to construct impact factors (Table 2.2). The use of surrogate taxa to reflect the overall response of biodiversity to environmental stress is questionable (Wolters *et al.* , 2006). In a global meta-study of multiple taxon responses to disturbance, Wolters *et al.* (2006) found a weak average correlation between taxa (r = 0.38). A number of factors influence this including habitat type, taxon, temporal and spatial scale. Yet precise roles of these factors are poorly investigated and unpredictable in novel situations (Wolters *et al.* , 2006). In the context of LCA, methods should prioritize major trophic or functional groups, taxa which are sensitive to the relevant pressures, and expand coverage based on data availability and feasibility.

The use of deductive methods can aid in overcoming both taxonomic and geographic limitations. Such an approach has been employed in studies such as the BioScore tool to model the response of indicator species to a range of pressures (Louette *et al.* , 2010); the Biodiversity Intactness Index to model the effects of land degradation (Scholes & Biggs, 2005); and the Mean Species Abundance as part of the GLOBIO3 model (Alkemade *et al.* , 2009).

2.6.2 Geographic coverage

The geographic coverage of methods in only two impact categories, water use and climate change, is global with respect to the terrestrial environment (Table 2.2). However, current climate change methods cover only 20% of the total terrestrial area across forest, arid/semiarid and mountain regions. Vulnerable terrestrial areas, such as islands and polar areas should be prioritized for future work. The methods in remaining impact categories were developed for use in specific regions or biomes. Detailed biodiversity data on the distribution of species across many taxa is incomplete on a global scale. Worldwide species richness and endemism data is available in equal-area grids and likely to be relatively robust to undersampling only for birds and plants (Orme *et al.* , 2006;

Kier *et al.* , 2005; also see IUCN global assessments of other taxa). For other taxa, checklists of predefined terrestrial and aquatic biogeographic regions and expert opinion (e.g. Olson & Dinerstein, 2002; Olson *et al.* , 2001; Spalding *et al.* , 2007; Abell *et al.* , 2008) have been used to map and assess biodiversity (Brooks *et al.* , 2006b).

Currently no methods exist to quantify aquatic habitat change in the context of land and water use. Meta-analysis or regional case-studies could guide the development of impact factors for lakebed, riverbed and seabed habitats across regions and climates, such as that pursued by the GLOBIO3 model for aquatic and marine environments (http://www.globio.info/). Turak *et al.* (2011) illustrates how an SAR-based approach can be adapted to freshwater habitats, using river length in place of area, to reflect a range of pressures. Global, spatially differentiated maps of pressures on marine ecosystems exist for 17 anthropogenic pressures at a grid resolution of 1 km^2 (Halpern *et al.* , 2008). For climate change, freshwater effects could be included by adopting a similar approach to Xenopolous *et al.* (2005). For marine climate change impacts, Halpern *et al.* (2008) provides a spatially resolved global map of climate change impacts on marine biodiversity including sea temperature rise, ocean acidification and UV radiation. This work also provides globally mapped impacts due to pollution runoff into marine waters, information that could be used to regionalize acidification and eutrophication. Ecotoxicity requires further semi-field research to verify the relationship between PAF and PDF across regions and environments (Posthuma & de Zwart, 2006; Snell & Serra, 2000).

2.7 Research outlook

Our review has illustrated the currently poor state of endpoint biodiversity modelling in LCA. The deficiencies across impact categories are not solely due to data limitations or even the inherent complexities of the element under study. They are also conceptual and methodological in nature. In order to meaningfully represent biodiversity in LCA, we present two broad recommendations for future research, presented in order of importance.

Fill the conceptual cracks. We see a need to first address the methodological shortcomings of current approaches. Clearly and explicitly defining PDF is an essential and urgently needed first step. Experimenting with non-linear, potentially unifying relationships (such as the SAR) when scaling impact factors represents another promising area of research that would eliminate the need to derive new impact factors for each scale.

Challenge data limitations. We have highlighted a wealth of data on the distribution of pressures (the basis of fate factors), their effects on various taxa (the basis of impact factors), and the global distribution of biodiversity either through raw data, models or surrogate indicators (the basis of characterization factors). Such data should be used to regionalize existing methods in order to capture a representative sample of the Earth's diverse terrestrial, freshwater and marine habitats, and to include a range of representative taxa. Following this, integrating new drivers and impact factors reflecting additional attributes of biodiversity could further improve the modelling of biodiversity loss in LCA.

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Chapter 3

Is there any empirical support for biodiversity offset policy?

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Abstract

Biodiversity offsets are seen as a policy mechanism to balance development and conservation goals. Many offset schemes employ habitat restoration in one area to recreate biodiversity value that is destroyed elsewhere, assuming that recovery is timely and predictable. Recent research has challenged these assumptions on the grounds that restoration implies long time delays and a low certainty of success. To investigate these assertions, and to assess the strength of empirical support for offset policy, we used a meta-analytic approach to analyse data from 108 comparative studies of secondary growth (SG) and old growth (OG) habitat (a total of 1'228 SG sites and 716 OG reference sites). We extracted species checklists and calculated standardized response ratios for species richness, Fisher's alpha, Sorenson and Morisita-Horn similarity. We modelled diversity change with habitat age using generalized linear models and multi-model averaging, correcting for a number of potential explanatory variables. We tested whether 1) diversity of passively and actively restored habitat converges to OG values over time, 2) active restoration significantly accelerates this process, and 3) current offset policies are appropriate to the predicted uncertainties and time lags associated with restoration. The results indicate that in the best case, species richness converges to OG reference values within a century, species similarity (Sorenson) takes about twice as long, and assemblage composition (Morisita-Horn) up to an order of magnitude longer (hundreds to thousands of years). Active restoration significantly accelerates the process for all indices, but the inherently large time lags, uncertainty and risk of restoration failure require offset ratios that far exceed what is currently applied in practice. Restoration offset policy therefore leads to a *net loss* of biodiversity, and represents an inappropriate use of the otherwise valuable tool of ecosystem restoration.

3.1 Introduction

Primary habitat loss, disturbance and fragmentation arguably represent the greatest immediate threats to the global persistence of biodiversity and provisioning of ecosystem services (MA, 2005c). The active restoration of plant and animal communities represents an important tool for managing and rehabilitating ecological systems (Benayas *et al.* , 2009). The field of restoration ecology has developed in recent decades to generate and apply knowledge on successional processes to assist the recovery of degraded or destroyed ecosystems to some suitable reference point (SER, 2004). While the goals of restoration generally focus on ecosystem resilience, structure and function (Suding, 2011), the choice of reference should favour old growth undisturbed vegetation, even though explicit biodiversity conservation goals concern a minority of cases (SER, 2004). More generally, the establishment of a "characteristic assemblage of species" is amongst the nine core success criteria suggested by the Society for Ecological Restoration, with the provisioning of habitat for specific (rare) species as context-dependent (SER, 2004).

Despite the widely acknowledged challenges in achieving a full recovery of the structure, functioning and composition of damaged ecosystems (e.g. Suding, 2011; Hilderbrand *et al.* , 2005; Maron *et al.* , 2010; Suding *et al.* , 2004; Woodcock *et al.* , 2011), policies that permit the compensated loss of natural habitat have multiplied internationally in recent years. A wide variety of schemes exist, including "conservation/biodiversity/endangered species banking" and "wetland mitigation" in the U.S.; "habitat compensation" in Canada; "Green offsets", "BioBanking" and the "BushTender/EcoTender" system in Australia; the "Habitat and Birds Directives" and "Natura 2000" in the E.U.; "biodiversity offsets" in South Africa, Uganda and Brazil; and the emergence of the international "Business and Biodiversity Offsets Programme" (ten Kate *et al.* , 2004).

Compensation mechanisms may be grouped into two classes, based on the policy goal. Compensation through *averted loss* of natural habitat results in a "compensated net loss" of biodiversity at the landscape level. In contrast, a "no net loss" and "net positive gain" of biodiversity results from compensation through habitat restoration, and is a defining goal of *biodiversity offsets*, which are specific policy tools designed to prevent a trade-off between development and conservation (BBOP, 2012). Although some authors have included "compensated net loss" under the definition of biodiversity offsets (e.g. Maron *et al.* , 2012), Bull *et al.* (2012) object on grounds that the defining criterion of true offsets is not met (i.e. no net loss). However, averted loss may be suitable where rates of background habitat loss are demonstrably high, and where no strong biodiversity protection legislation or mandatory compensation policy exist (Gibbons & Lindenmayer, 2007; Maron *et al.* , 2012). Otherwise, the site where habitat loss has been "averted" would not be threatened (no additionality), and would itself require compensation if cleared, leading to a logical paradox (Maron *et al.* , 2012). Therefore, in the majority of regions where offsets are ingrained in policy (mainly developed countries), they *should be* fundamentally defined by the use of ecological restoration as a compensation mechanism to ensure at least a no net loss of biodiversity (Bull *et al.* , 2012). Despite this, and to avoid possible confusion, we use the term *restoration offsets* throughout the text to specifically refer to the use of habitat restoration as a compensation mechanism.

Restoration offsets aim to recreate habitat in one area, the "offset site", in order to compensate the development-driven loss of habitat in another area, the "impact site" (McKenney & Kiesecker, 2010; Maron *et al.* , 2012). The ratio of biodiversity value gained to value lost is usually quantified using "offset ratios", based on some form of integrative habitat quality or service index (Parkes *et al.* , 2003; McCarthy *et al.* , 2004), which is then converted to habitat area ratios (i.e. hectares gained to hectares lost). Offset ratios may, but usually do not, include additional corrections ("multipliers") for time delays in the gain of biodiversity value, future uncertainties, the risk of restoration failure and the spatial configuration of habitat in the landscape (Bruggeman *et al.* , 2005; Moilanen *et al.*

, 2009a; Bekessy *et al.* , 2010; Overton *et al.* , 2013). Offsets are also usually employed in areas of high conservation value where development is deemed unavoidable, specifically addressing the loss of old growth or remnant vegetation, or habitat for specific species (e.g. "banks" of high-conservation value habitat in the U.S., native remnant vegetation in Australia or the "Natura 2000" network of Special Areas of Conservation of in the E.U.).

Lauded for their purported economic efficiency (ten Kate *et al.* , 2004), offsets have been severely criticized in recent years by conservationists, ecologists and restoration practitioners (e.g. Gibbons & Lindenmayer, 2007; Walker *et al.* , 2009b; Maron *et al.* , 2010). The main criticisms relate to serious doubts that restoration ecology, a young and untested field, can deliver what is required of biodiversity offsets: an exact recreation of impacted old growth biodiversity value within a reasonable time period and acceptable risk of failure (Bekessy *et al.* , 2010; Maron *et al.* , 2012). Time lags between the loss and gain of suitable old growth habitat, uncertainty over whether conditions for successful restoration will exist in the future (e.g. bottlenecks in ecological resources), the risk of outright restoration failure, and inadequate measurability of the biodiversity value that is lost or recreated are amongst the most serious concerns (Maron *et al.* , 2012). In order to confirm or dispel these criticisms, adequate information about the biodiversity value of restored habitat, estimates of uncertainties and failure rates of restoration projects is required. The current evidence base pertaining to these factors is patchy and poorly investigated. Only a few recent studies have attempted a quantitative analysis of biodiversity change in passively and actively restored ecosystems (Dunn, 2004; Chazdon *et al.* , 2009a; Jones & Schmitz, 2009; Benayas *et al.* , 2009; Dent & Wright, 2009). A comprehensive statistical model of biodiversity change following ecosystem damage and restoration is currently lacking. Comprehensive in this sense refers to accounting for potential explanatory factors other than habitat age (e.g. ecosystem type, biogeographic and landscape context, taxon), addressing the common pitfalls of pseudo-replication (Ramage *et al.* , 2012) and undersampling (Gardner *et al.* , 2007a), and quantifying the benefits of active habitat restoration in terms of accelerated biodiversity recovery (Suding, 2011).

In this study, we construct such a model using a meta-analytic approach with the overall aim of providing a stronger evidence base for a more informed debate on offsets. We also investigate the limits and opportunities of restoration ecology as a tool to recover lost biodiversity value. Specifically, we address five main hypotheses, paying particular attention to assessing whether the conditions are met for successful restoration offsets, in terms of robustly preventing a net loss of biodiversity (hypotheses 1–3). We also quantify the influence of two common methodological pitfalls in comparative biodiversity studies that rely on the "before-after/control-impact" (BACI) design (hypotheses 4 and 5).

Hypotheses and expectations:

1. *Species diversity indicators (e.g. species richness, diversity and similarity indices) are initially impacted by disturbance, and converge to old growth reference values over time* (convergence hypothesis). Indicators of species similarity and assemblage composition are expected to show increasing similarity between secondary growth (SG) and old growth (OG) sites with habitat age. This relationship should persist when isolated from other explanatory factors such as taxon, ecosystem type, site disturbance history, patch size, connectivity etc.

2. *Active habitat restoration speeds up this process relative to natural succession, also termed passive restoration* (restoration hypothesis). The time required for diversity indices to reach OG reference levels is expected to be notably reduced under a regime of active restoration.

3. *Biodiversity offset policy and ratios are consistent with the uncertainties and time lags associated with habitat restoration* (offset ratio hypothesis). The range of offset ratios employed in practice is expected to roughly match the requirements of "robustly fair offsets" (Moilanen *et al.* , 2009a) based on our empirically modelled recovery times and restoration failure rates. For example, a 150 y time horizon, 50% failure rate and variable uncertainty and time discount rate leads to offset ratios ranging 18 to 95 units gained per unit lost (Moilanen *et al.* , 2009a).

4. *A higher intrinsic species richness in OG habitat leads to a systematic undersampling bias (i.e. not recording all species at a site) relative to SG habitat, leading to biased biodiversity comparisons* (undersampling hypothesis). Proxies of sampling effort and sampling completeness should correlate with differences in diversity indices between habitats (Gardner *et al.* , 2007a).

5. *A higher intrinsic rate of spatial species turnover in OG habitat, relative to SG habitat, leads to biased biodiversity comparisons when site replication or spatial scale of sampling is limited* (sampling scale hypothesis). OG habitat is expected to exhibit higher levels of spatial species turnover relative to SG habitat at different stages of growth (Gardner *et al.* , 2007a).

3.2 Methods

3.2.1 Literature search

We searched for available studies that compared the species diversity of secondary growth (SG) and old growth (OG) habitats. We started by assembling previous quantitative reviews of the secondary growth and restoration ecology literature with a global focus on terrestrial habitats (Brown & Lugo, 1990; Houerou, 2000; Dunn, 2004; Bowen *et al.* , 2007; Gardner *et al.* , 2007a; Liebsch *et al.* , 2008; Chazdon *et al.* , 2009a; Jones & Schmitz, 2009; Benayas *et al.* , 2009; Dent & Wright, 2009). We screened the reference list of these reviews for studies that passed the following three selection criteria: (i) sampling data from SG habitat and a comparable OG reference, (ii) known age of the SG habitat, and (iii) species abundance or occurrence data for both habitats. We further conducted a Google Scholar and ISI Web of Science search with relevant keywords drawn from the literature (Appendices, "Expanded methods"). In a final step, we wrote to the correspondence email address of all potentially suitable studies conducted after 1990, that did not present quantitative diversity data, and requested site-level species data and additional information pertaining to our hypotheses (e.g. sampling protocol, site history, landscape factors).

3.2.2 Data extraction

We extracted species data from each study based on reported species lists, either aggregated across habitats (i.e. collection of OG/SG sites) or at the sampling site level when available. These data were composed of species occurrences, relative abundances, or individual counts. For count data, we calculated both observed species richness and indicators of alpha diversity that are assumed to be robust to undersampling bias: Fisher's alpha (Fisher *et al.* , 1943) and the species richness estimators Chao 1, Jackknife 1 & 2 and the Abundance-based Coverage Estimator (ACE), which have been shown to perform well under conditions of undersampling (Brose & Martinez, 2004; Beck & Schwanghart, 2010). For relative abundance and incidence data without replication, only uncorrected indices could be calculated. To measure pairwise compositional similarity between SG and OG samples, we used two common similarity metrics: the occurrence-based Sorenson index, which was applicable to the entire dataset, and the Morisita-Horn index, which could only be calculated for count and relative abundance data (see Appendices, "Expanded methods" for formulae of indices).

We extracted predictor variables from each study relating to ecological and methodological factors relevant to our five hypotheses (convergence, restoration, offset ratios, undersampling, and scale hypothesis). We classified SG habitat types into those resulting from *passive* or *active* restoration. Passive restoration relates purely to the cessation of disturbance without any additional remedial human activity (Suding, 2011). This typically results in recovery through natural processes following land abandonment, shifting cultivation, logging, fires, exclusion of grazing etc. Active restoration involves a "guided recovery" through targeted human interventions such as vegetation planting (e.g. with native or exotic species and under mixed or monoculture regimes), animal reintroductions, and the storage and replacement of topsoil following surface mining

(Suding, 2011). Beyond this simple active/passive classification, the literature was too sparse to facilitate a more detailed investigation of different restoration techniques.

We also extracted ecological and methodological data, consisting of a simplified biome classification (1: coniferous forest, 2: moist broadleaf forest, 3: transitional dry forest/woodland, 4: open shrub/grassland vegetation), realm (1: Australasian, 2: Afrotropic, 3: Indo-Malay, 4: Nearctic, 5: Neotropic and 6: Palearctic; Olson *et al.* , 2001), taxon (1: trees, 2: other mainly non-woody plants, 3: birds, 4: mammals, 5: herpetofauna, 6: insects and 7: other invertebrates), elevation (masl), absolute latitude (distance from the equator in degrees), patch size (increasing logarithmic size classes of 1: $<$ 10 ha, 2: 10–100 ha, 3: 100–1000 ha and 4: $>$ 1000 ha), patch connectivity (0: isolated, 1: connected), distance between sampling sites (km), past disturbance intensity (1: extensive transformation [ET], 2: extensive occupation [EO], 3: intensive transformation [IT] and 4: intensive occupation [IO]). Our past disturbance intensity classes describe the magnitude (extensive/intensive) and duration (transformation/occupation) of anthropogenic disturbance before the onset of passive or active restoration. Due to a lack of data on ecotoxic (e.g. pesticide use), eutrophic (fertilizer use) or biotic (e.g. herbivore density, introduced species) pressures, we applied a structural definition of disturbance intensity similar to Dent and Wright (2009). For example, extensive disturbance of forest habitat included selective logging, agroforestry and shifting agriculture, whereas intensive disturbance included conversion to agriculture, pasture or clear-cut logging. Occupation and transformation relate to whether the land was disturbed then immediately left to recover, or whether there was an occupation phase that prevented recovery. A more detailed description of each disturbance class is contained in the supporting information (Appendix A.8.1).

We addressed unequal sampling design using a relative index of the difference in sampling effort between habitats based on plot size, within-site spatial and temporal replication, and the number of sites per habitat. We also included the average number of sites in the comparison as a measure of sampling scale (assuming that more sites indicates a larger spatial scale of sampling). Habitat edge, fragmentation and landscape matrix effects were ignored due to a lack of data. A complete description of the diversity and predictor data is provided in the Appendices (Appendix 8.2.1).

3.2.3 Response ratio

For all possible within-study SG–OG sample combinations, we constructed a $\log_{10}$-transformed response ratio (RR; Hedges *et al.* , 1999; Borenstein *et al.* , 2009) for each diversity index:

$$RR_i = log_{10}\left(\frac{x_{i,SG}}{x_{i,OG}}\right) \tag{3.1}$$

Where i identifies the variable of interest and x_i is its value in either SG or OG habitats. The response ratio represents a standardized effect ranging from $-\infty$ to $+\infty$, where negative values indicate a *lower* value in the SG sample, positive values indicate a *higher* value in the SG sample, and 0 signifies no difference. For pairwise similarity between samples, the Sorenson and Morisita-Horn indices already represents a response metric between two samples (i.e. proportional similarity). However, raw similarity indices are unstandardised because beta diversity change may occur at different scales in different studies, due to background patterns of compositional change varying across taxa, spatial scales and ecosystem type, among other factors (Dent & Wright, 2009). Therefore a subset of studies with true replication in OG habitat was used to construct a standardized response ratio for the two similarity indices. This involved expressing SG–OG similarity in each study as a proportion of average "background" similarity in OG control comparisons:

$$RR_{SIM} = log_{10}\left(\frac{SIM_{SG\text{–}OG}}{SIM_{OG\text{–}OG}}\right) \tag{3.2}$$

Where $SIM_{OG\text{–}OG}$ is the average value across all OG comparisons in the study.

3.2.4 Generalized Linear Models (GLMs)

To assess the importance of influential factors acting on the age-diversity relationship (i.e., identifying the most important predictors in a statistical model), a series of generalized linear models (GLMs) were constructed using various subsets of the full dataset (Gaussian error distribution on log-transformed response ratio). First, the data were sub-set according to data availability for three broad groups of predictors. This led to three separate analyses, with analysis 1 focusing on patch dynamics and titled *patch model* throughout the text, analysis 2 assessing the influence of distance between samples and referred to as *distance model* in the text, and analysis 3 quantifying the influence of general anthropogenic (e.g. disturbance intensity), ecological (e.g. taxon) and biogeographic factors (e.g. realm), and referred to as the *general model* in the text. A detailed description of the predictors in each analysis is provided in the Appendices ("Expanded methods" and Table A.8.1). The *patch* and *distance* models were constructed only for similarity indices, whereas the general model was constructed for all diversity indices.

To avoid pseudo-replication and bias induced by the clustering of data within individual studies (e.g. Ramage *et al.* , 2012), resampling of the full dataset was conducted throughout all analyses. Unless otherwise stated, this involved randomly selecting a single standardized SG–OG sample comparison per study. Each model (patch, distance and general) involved resampling the full dataset 10’000 times with replacement. A “full” GLM (i.e. including all predictor variables) was fitted to each resampled dataset (sample size equalled the number of studies included in the particular model). These full models were each subjected to a model selection and coefficient averaging algorithm using the second-order bias corrected version of the Akaike Information Criterion (AICc) adapted for small samples (Burnham *et al.* , 2011). Model weights were developed based on their AICc difference to the best model: $w_i = exp(-(AICc_i - AICc_{best}))$, where w_i is the weight of model i, which compares its AICc value, $\mathrm{AICc_i}$, to that of the best model, $\mathrm{AICc_{best}}$ (Calcagno and de Mazancourt 2010).

Model averaging (Burnham & Anderson, 2002) was performed across the 100 best models for each of the 10’000 resample runs, using model weights (see above) to generate importance values for predictors (AICc-weighted proportion of models in which they occurred) and estimates of coefficients (AICc-weighted average of coefficient values). This produced a set of averaged parameter coefficients, unconditional coefficient variances and importance values for each of the 10’000 model runs. The percentage of deviance explained (%DE) of the full model and best model for each of these 10’000 runs was also recorded. This entire protocol was repeated for each of the data subsets used for the three models (i.e. patch, distance and general models using the relevant diversity indices).

3.2.5 Recovery speed and probability of restoration failure

To visualize recovery trajectories in relevant diversity indices, parameters were used from the general model to plot the predicted relationship between age and diversity under the partial effects of the other predictors (i.e. plotting the age–diversity relationship using the age coefficient, and changing taxon, latitude, restoration method etc.). We filtered the 10’000 model selection runs to remove those with a poor predictive ability, arbitrarily defined as leading to a best model with a deviance explained value of $< 10\%$. Additionally, we removed models where the age effect was negative, as these runs can immediately be disregarded as indicative of a lack of convergence to OG biodiversity values, as posited by hypotheses 1 (convergence hypothesis). We interpreted the proportion of model runs removed as a proxy indicator of restoration failure rates (i.e. data that show a negative or no trend with age).

Parameters of each of the remaining selection runs were then weighted by their importance values, and inserted into a linear model formula (“full model averaging”; Lukacs *et al.* , 2010). The weighing step scales the contribution of predictors according to their importance, but retains information from all potentially suitable candidate models and predictor variables (Symonds & Moussalli, 2011). We estimated recovery times, defined as the time needed for average diversity values in SG habitat to return to reference values, taking the average OG–OG value per study, and the average minus one standard deviation to reflect variability. In a sensitivity analysis step to test second hypothesis (restoration hypothesis), that active restoration substantially accelerates the recovery process, we developed 28 predictor combinations of taxonomic group, realm and latitude, and repeated these for both passive and active restoration. For these recovery predictions, we used a recovery criterion of within $\frac{1}{2}$ a standard deviation of the OG–OG reference to be precautionary. Where predictions covered a long time horizon, this tentatively assumes that linear trends observed within the data range of a few hundred years continue into the distant future. This entire procedure generated hundreds of individual recovery trajectories, allowing uncertainties to be explicitly quantified and visualized.

3.2.6 Undersampling bias and spatial sampling scale

The magnitude of undersampling bias was assessed by calculating the completeness of sampling as the ratio of observed and estimated species richness values for sampling sites in both habitats. Differences in sample completeness between paired SG and OG sites were tested for significance using both a parametric paired t-test and non-parametric Wilcoxon rank sum test (using resampled data to account for pseudo-replication). We quantified the potential influence of spatial sampling scale on SG and OG diversity comparisons indirectly, by quantifying spatial species turnover using Whittaker's beta (Whittaker, 1972). The spatial scale of sampling will influence the results of diversity studies if habitats being compared exhibit differing rates of spatial species turnover (Gardner *et al.* , 2007b). We assumed that the number of sites per treatment was correlated to the spatial scale of sampling, and quantified the average distance between sites in the dataset to offer an estimate of the scale to which our findings apply (see results). Whittaker's beta was calculated at the 3-site scale for studies with multiple replicates per treatment and site-level species data (i.e. all possible combinations of 3 sites within a habitat and age class). We computed Whittaker's beta for aggregated age classes consisting young SG ($<$ 15 y), mature SG ($>$ 15 y), OG habitat, and combined pools of (young and mature) SG and OG samples. Beta diversity was visualized using box and whisker plots, and tested for significance using an ANOVA.

All statistical analyses were performed using the programming language R vers. 2.15.1 (http://cran.r-project.org/). The study made extensive use of the packages "fossil" and "vegan" for calculating diversity indices, "MASS" and "glmulti" for modelling and model selection, and "ggplot2" and "corrgram" for plotting and visualization.

3.3 Results

3.3.1 Search results

The search resulted in 101 studies that passed the selection criteria. The authors of seven additional studies responded to requests for disaggregated data (Parrotta & Knowles, 2001; Abbott *et al.* , 2003; O'Dea & Whittaker, 2007; Aerts *et al.* , 2008; Bihn *et al.* , 2008; Bowen *et al.* , 2009; Costa *et al.* , 2010). 42 studies presented replicate OG data (Appendices, Table A.8.2). The final dataset contained information from 1'228 SG sites of different ages and 716 OG reference sites, giving a total of 7'954 within-study comparisons (both SG–OG and OG–OG). Of these, 7'597 comparisons could be standardized to background OG–OG variability in species similarity (i.e. they originated from the 42 studies presenting OG replicate information). The standardized data covered 6 of the 7 realms of Olson *et al.* (2001), extending from latitudes 43°S to 63°N (Figure 3.1), spanning 1 to 140 years post-disturbance (non-standardized data extended to 235 years; Appendix A.8.1). This dataset was sub-set for the three principal GLM analyses – patch, distance and general models – with the different diversity indices (Table 3.1). A summary of the entire dataset (i.e. taxonomic and geographic representation) can be found in the Appendices (Appendix A.8.2).

3.3.2 GLM parameters and predictions

The GLM runs resulted in model-averaged importance values and coefficients for predictors differing considerably across diversity indices. Correlation between variables included in the model runs was generally low (< 0.4). Only latitude and altitude exhibited a (negative) correlation coefficient that exceeded 0.7 (0.90 in the general model 3; Appendix A.8.2). The average structural goodness of fit (percentage of deviance explained) of the best models from the selection runs was generally low, ranging ca. 10% to over 40%, depending on model and diversity index (Table 3.1). The response to age was not uniform, and exhibited a strong and important effect only for the Sorenson and Morisita-Horn index. Alpha diversity indices (species richness and Fisher's alpha) exhibited a weaker link to age compared to other predictors (Appendix A.8.5). Previous disturbance intensity and sampling effort were of particular importance for species richness. Fisher's alpha exhibited a strong relationship to realm, latitude and sampling effort, and had a neutral intercept, indicating no particular direction of change due to disturbance.

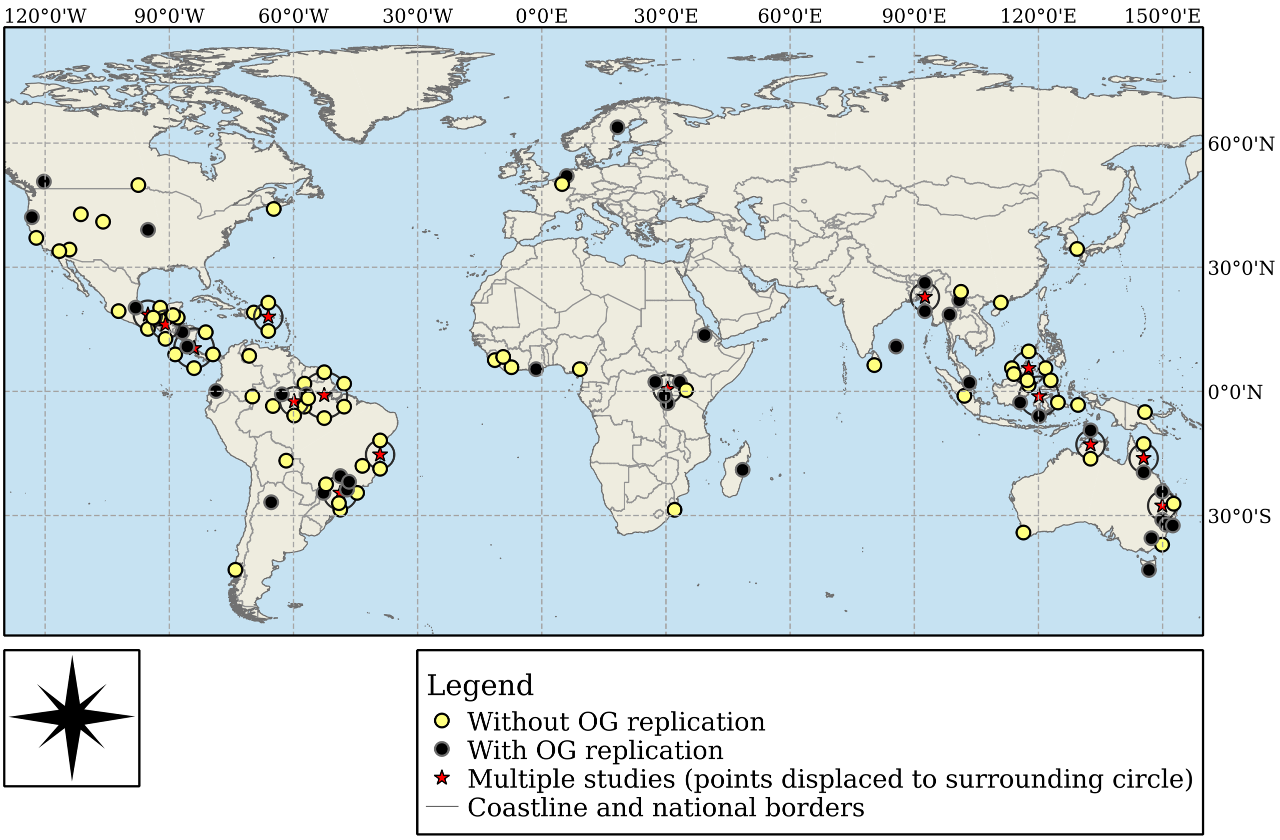

Figure 3.1 – Geographic location of studies included in the meta-analysis. Geographic location of studies included in the meta-analysis, showing those presenting replicated old growth (OG) data (solid, black) and those lacking OG replication (hollow, yellow). Localities featuring multiple studies represented by red star (with study symbols displaced to surrounding circle).

Figure 3.2 presents parameters of model 3, the general model, for Sorenson and Morisita-Horn response ratios. Parameters of models 1 and 2, which investigated the effects of patch dynamics and distance between samples, respectively, are only discussed in the text, with details in the Appendices (Appendices, Figures A.8.3 and A.8.4). Because of limited data, we used a simplified biome (forest, non-forest) and taxonomic classification (plants, birds, mammals, herpetofauna and invertebrates) for both patch and distance models. The patch model showed generally weak support for almost all predictors. For Sorenson similarity, taxon, SG connectivity and OG patch size had the highest importance, with the latter two predictors both showing an importance-weighted positive effect on similarity of magnitude 2.6% and 0.6% per increment, respectively. For the Morisita-Horn index, only SG patch size was moderately important (34%), and had a negative effect on similarity of magnitude 3.8% per size increment (Appendices, Figure A.8.3).

Model 2 incorporated the influence of distance on ecological similarity. Where data was available, an average distance of 17.77 km was observed (standard deviation of 30.64

Diversity index	**# Stud.**	**# Comp.**	**Sverage %DE of best models**		
			All runs	**%DE > 10**	**%DE > 10 & age +ve**
1. Patch model					
Sorenson	18	2070	42.29	NA	NA
Morisita-Horn	16	2016	25.18	NA	NA
2. Distance model					
Sorenson	25	4773	15.22	NA	NA
Morisita-Horn	21	4618	10.32	NA	NA
3. General model					
Sorenson	39	5019	9.15	21.47	20.74
Morisita-Horn	34	4855	17.99	35.46	32.67
Species richness	106	5410	12.27	16.39	16.06
Fisher's alpha	72	4749	25.50	28.78	29.22

Table 3.1 – Summary information of statistical models. Table shows model fit statistics for three statistical models ("patch", "distance" and "general" models) for all diversity indicators. In general, model fit (% deviance explained) was low, ranging ca. 10%–40% depending on model and diversity index. However, filtering out poorly performing models, and those with a negative age effect improved statistical fit. Within-study SG-OG comparisons were resampled with replacement 10'000 times, taking one comparison per study (n = sample size), and model selections performed on each resampled dataset. Percentage deviance explained (%DE) of best models given for all resample runs; resample runs with best model %DE > 10; and best model %DE > 10 and age effect positive; NA = Not applicable

km). A characteristic negative coefficient for distance was observed for both indices, confirming the presence of distance decay in ecological similarity (Soininen *et al.* , 2007). A dataset-average halving distance of ca. 656 km was observed for the Sorenson index (i.e. the distance required to reduce similarity by 50% from its starting value, calibrated at 1 km with dataset-average values for all other predictors). This compares remarkably well to Soininen *et al.*'s (2007) global average halving distance of 639 km for comparable occurrence-based similarity indices. In general, distance was of a similar importance to age for both Sorenson and Morisita-Horn response ratios (Appendices, Figure A.8.4), but the effect was weak, predicted to exert a minimal effect on the data (at the dataset average distances of 17.46, standardized Sorenson and Morisita-Horn similarity would be reduced by less than 0.3% and 0.05%, respectively).

Focusing on model 3 (the general model), taxonomic effects were unimportant and relatively weak for Sorenson similarity (Figure 3.2a). Conversely, taxon was a strong predictor of Morisita-Horn similarity, indicating a negative coefficient for mammals, insects and herpetofauna (lower SG–OG similarity, slower recovery), and near neutral coefficients for plants, invertebrates and birds (Figure 3.2b). A larger dataset for Sorenson similarity allowed woody-plants (trees) to be partly isolated from non-woody and shrub species, revealing a negative coefficient for trees (Figure 3.2a). The effect of differences in sampling effort between SG and OG habitat was weaker for similarity indices than for alpha diversity. A moderately important positive effect for Sorenson similarity indicates that disproportionate sampling effort in SG habitat to a ratio of 10:1 would correlate with an *increase* in measured SG–OG similarity of about 1.2% (0.5% for Morisita-Horn). A similar effect was observed for the number of sites in the comparison (a proxy for the scale of sampling), which is predicted to increase similarity by 0.5% and 0.7% for Sorenson and Morisita-Horn, respectively, when 10 additional sites are sampled in each habitat. The effect of biome type differed between the indices, with a non-forest (open) biome predicted to have a marginal positive effect on Sorenson similarity (of about 0.2%), but no notable effect on Morisita-Horn. The effect of latitude and elevation differed between the indices (negative for Sorenson, slightly positive for Morisita-Horn). Active restoration was more important for Sorenson similarity, and predicted to have a positive effect of about 1.2% (0.1% for Morisita-Horn). While the above effects may appear weak in terms of absolute changes in percentage similarity, they must be viewed in relation to the intercepts of the models, which indicated a global decrease in Sorenson and Morisita-Horn similarity of 18.8% and 31.7%, respectively, relative to OG reference values (Figure 3.2).

Hypothesis 1: SG and OG diversity converges over time

We filtered the 10'000 model selection runs to remove the worst performing models (%DE > 10 criterion, see Methods section "*Recovery speed and probability of restoration failure*"). This resulted in the removal of 37.4%, 13.5%, 67.2%, and 51.4% of selection runs for species richness, Fisher's alpha, Sorenson and Morisita-Horn similarity, respectively. The proportion of remaining runs that supported a positive effect of age (supporting the convergence hypothesis) was 65%, 87.9% and 61.1% for species richness, Sorenson and Morisita-Horn, respectively (we omitted Fisher's alpha from this and following analyses because of the very weak link to age and largely neutral intercept; Appendices, Figure A.8.5). The data therefore support the hypothesis 1 (convergence hypothesis), that SG and OG diversity converges over time. The age effect was significant for all three diversity indices (proportion test, $n = 6269$, $p < 0.001$ for species richness; $n = 3480$, $p < 0.001$ for Sorenson; $n = 4860$, $p = 0.062$ for Morisita-Horn).

Hypotheses 2: Active restoration accelerates recovery

The filtered model runs (%DE > 10 and positive age coefficient) were used to estimate recovery times for various combinations of model parameters (scenarios) under regimes of both passive and active restoration (Figure 3.3). We plotted a selection of these model combinations for species richness (Appendices A.8.6), Sorenson (Figure 3.3a) and Morisita-Horn similarity (Figure 3.3b). The full range of recovery time predictions (i.e. to within $\frac{1}{2}$ a standard deviation of the OG reference) is given in the supporting information (Appendices, Tables A.8.4, A.8.5 and A.8.6). Plots of the filtered model

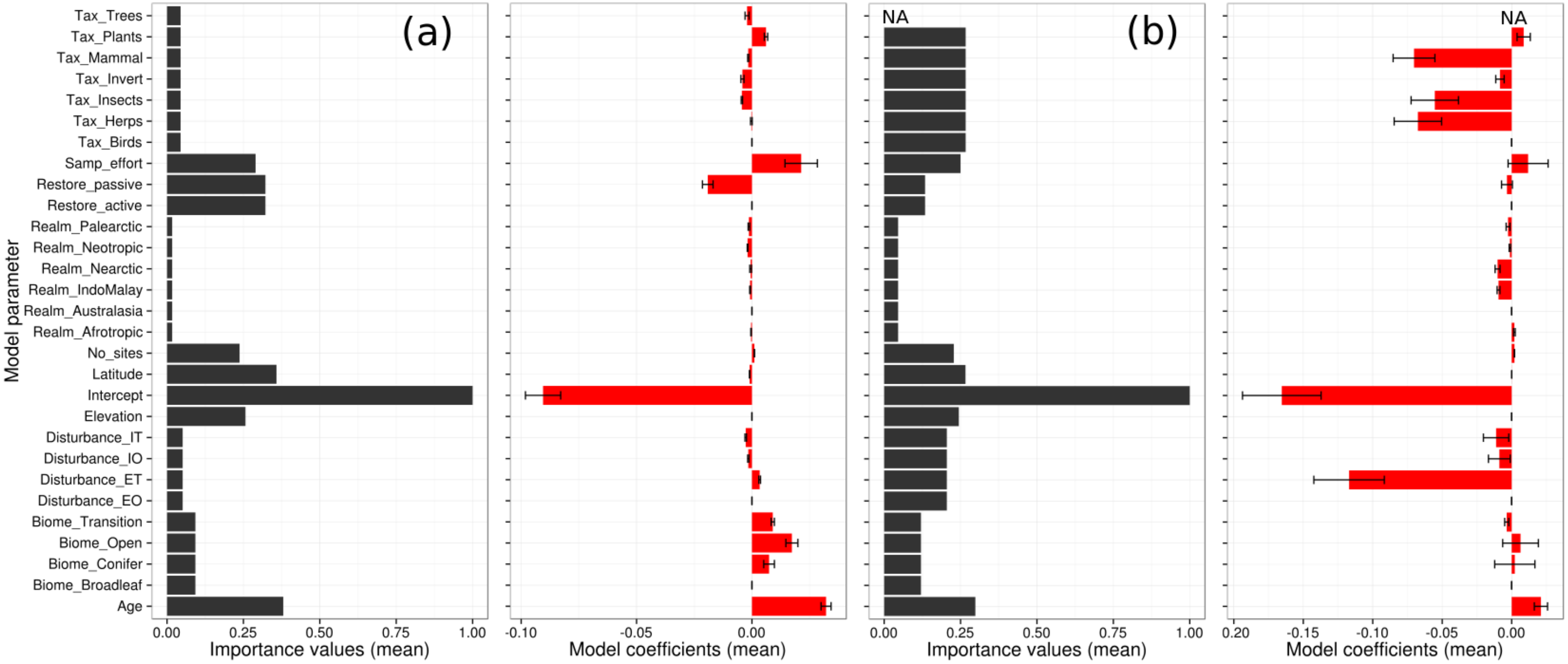

Figure 3.2 – **Parameters of the general recovery model**. Model-averaged importance values (left) and coefficients +/- variance (right) for the "general model" of Sorenson (a) and Morisita-Horn (b) similarity, quantifying the effects of a range of ecological and anthropogenic influences on the recovery process. The y-axis lists predictors tested in the model for an influence on ecological similarity between sites. Age was the most important predictor for both indices, but environmental (elevation and latitude) and methodological (sampling effort and site replication) factors, as well as restoration method also showed varying and often equally important effects. Derived from N = 10'000 resampled model runs with replacement and sample sizes (# studies) of n = 39 (Sor) and n = 34 (M-H). NA = Not available

regressions illustrated high uncertainties across the model runs for all indices (Figure 3.3 and Appendices, Figure A.8.6). We emphasize that these trajectories represent overly certain estimates under overly optimistic conditions, effectively representing the best case scenario (i.e. when recovery actually occurs).

Average values across the various realms and taxa indicate passive recovery of species richness occurs within about a century in forested biomes (median = 73.5 y, range = 46.7–138.8 y) and within about a decade in non-forest (open) biomes (median = 7.5 y, range = 4.7–14 y). Recovery of Sorenson similarity is predicted to take about twice as long, between one and two centuries in forested habitats (median = 144.1 y, range = 105–230.1 y) and about half a century in non-forest biomes (median = 97.2 y, range = 71.4–155.2 y). In contrast, Morisita-Horn predictions range about an order of magnitude larger, with forest biomes predicted to take over a millennium (median = 1'644 y, range = 218–17'024 y) and non-forest biomes about 800 y (median = 792 y, range = 105.5–8'209 y). Active habitat restoration was predicted to accelerate the recovery of species richness by 80%, Sorenson similarity by about 60%, and Morisita-Horn similarity by about 15%, indicating often considerable benefits across diversity indicators, supporting our restoration hypothesis (2).

Hypothesis 3: Offset policy accounts for time lags and uncertainties

We used 100 years as a generous upper limit for "successful restoration" in the context of restoration offsets. This is due to the fact that robust offset ratios increase rapidly with increasing time-lags, uncertainty and risk of restoration failure (Moilanen *et al.* , 2009a; Maron *et al.* , 2012). Based on the individual model runs used to calculate our scenario predictions (Appendices Tables A.8.4 to A.8.6—showing mean results only), active restoration significantly increases the frequency of success for all three diversity indices, when compared against passive restoration (Table 3.2; Wilcoxon rank sum test on paired data, $n = 154$, $p < 0.001$). In absolute terms, successful active restoration occurs within the data at a probability of 0.58 for species richness (forest = 0.56, non-forest = 0.68), 0.56 for Sorenson similarity (forest = 0.54, non-forest = 0.59), and 0.29 for Morisita-Horn similarity (forest = 0.30, non-forest = 0.29; Table 3.2).

A window of opportunity for restoration offsets can be generated by combining these figures with the frequencies of restoration failure calculated previously (i.e. the proportion of model runs supporting a negative age effect). The results suggest that restoration offsets would generally meet the target of a "no net loss" of diversity (within 100 y) in about 40% of cases for species richness (forest = 36.8%, non-forest = 44.5%), 50% for Sorenson similarity (forest = 49.3%, non-forest = 57.2%), but only 18.9% for Morisita-Horn similarity (biome independent). If all biodiversity features must successfully recover

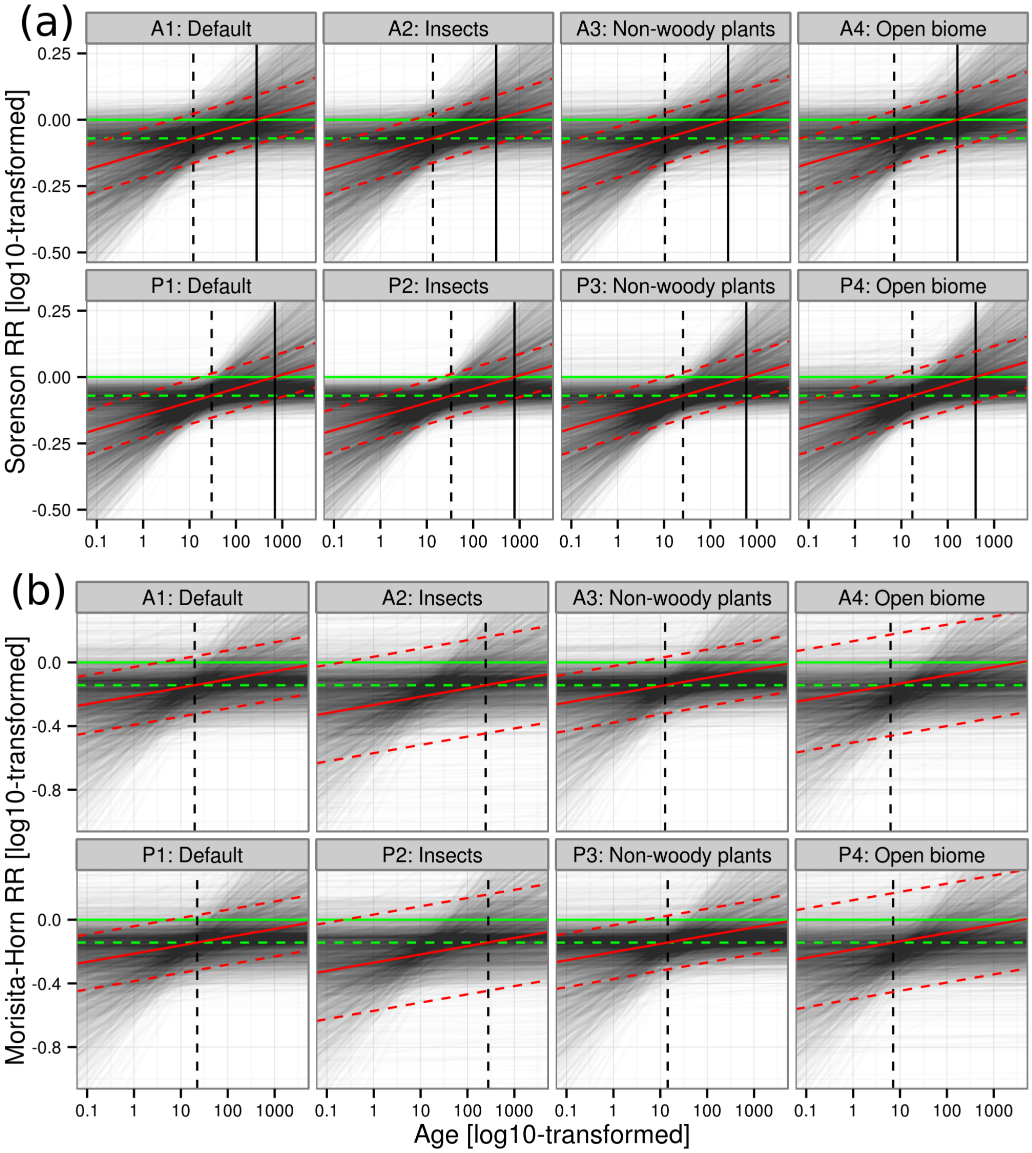

Figure 3.3 – **Recovery trajectories**. Modelled recovery trajectories for Sorenson (a) and Morisita-Horn similarity (b) based on partial effects of selected predictors for active restoration (A1-4, upper row) and passive restoration (P1-4, lower row). Both age (x-axis) and the response ratio (y-axis) are $\log_{10}$-transformed, with a response-ratio value of zero indicating no difference in standardized similarity between SG and OG samples (i.e. full recovery). Regression lines plotted separately for each resample model run. Line density shading = heat map where darkest shading (black) indicates that at least 2.5% of regression lines overlap. Mean coefficient value and +/- 1 s.d. shown as solid and broken red lines, respectively. Solid vertical black line indicate intersection points between the average slope and zero on the y-axis (i.e. the "recovery time") and dotted black lines indicate intersection with 1 s.d. of the OG reference. Facets represent changes to individual parameters of the default "general model", which assumes taxon = birds, biome = broadleaf forest, realm = Australasia, disturbance = extensive occupation, with mean dataset values for continuous variables.

(i.e. exact equivalence across all three indices), and the proportion of success is used as a measure of average biodiversity value in the impact site, this implies simple offset ratios of 1/0.189 = 5.29 units gained per unit lost, with no uncertainty or time discounting (modified from eq. 1 in Moilanen *et al.* 2009). If we assume that the average standard deviation of the intercept in our models reflect the error weight of the diversity of the restored site ($w = 0.144$ in eq. 2, Moilanen *et al.* , 2009a), the robust ratio increases to 1/(0.164-0.144) = 22.22 units gained per unit lost. To provide an indication of the additional effect of time discounting, applying an exponential model with a 4% discount rate (Overton *et al.* , 2013) to recovery times of 50, 75, and 100 y leads to multipliers of 7.7, 21.4 and 59.3, respectively.

Although these represent simplifications of the models of Moilanen *et al.* (2009), it clear that our results imply relatively high offset ratios, ranging perhaps 10–100, depending on which corrections are included. Viewed in context, even the largest existing offset ratio of 30:1, applied to critically endangered ecosystems in South Africa's Western Cape offset policy (DEADP 2007) may be insufficient to prevent a net loss of biodiversity in the medium to long run.

Hypotheses 4 & 5: Undersampling and inadequate sampling scale undervalue OG habitat

There was no sign of a systematic bias in undersampling towards either SG or OG habitat. The estimated sample completeness across studies was similar for SG and OG habitats (Figure 3.4a & 3.4b), with both median values close to 80%. The general pattern in paired SG–OG samples across studies was idiosyncratic, with higher undersampling observed in both habitats depending on the context (Figure 3.4c). Difference in sample completeness in all paired single-site samples indicated that species inventories in SG habitat are 2–3% less complete than OG habitat (based on median values represented by the vertical dotted lines in Figures 3.4a and 3.4b). However this difference was not significant according to both resample-based Wilcoxon rank sum test (100 resamples, n = 150, average p-value = 0.259) and Student's t-test (100 resamples, n = 150, average p-value = 0.129). Although undersampling appears to be prevalent across all studies, and is of considerable magnitude in many cases, the general trend is highly variable and unlikely to exert a serious bias toward one habitat type relative to the other. We quantified spatial species turnover in each habitat using Whittaker's beta. OG habitat had a higher rate of turnover than mature SG habitat, and was roughly equal to young SG (Figure 3.4d). The combined samples of OG + young SG increased Whittaker's beta by about 5%, indicating that the SG habitat contributes early-successional species to the combined species pool (which was

Diversity index	**Proportion of model runs**						
	%DE $>$ 10	**Age +ve**	**%DE $>$ 10 & age +ve**	**Passive success, 100 y – forest**	**Active success, 100 y – forest**	**Passive success, 100 y – non-forest**	**Active success, 100 y – non-forest**
Species richness	0.626	0.664	0.407	0.499 (± 0.018)	0.566 (± 0.011)	0.622 (± 0.010)	0.684 (± 0.011)
Fisher's alpha	0.865	0.816	0.718	NA	NA	NA	NA
Sorenson	0.348	0.824	0.306	0.365 (± 0.039)	0.539 (± 0.035)	0.419 (± 0.039)	0.586 (± 0.032)
Morisita-Horn	0.486	0.625	0.297	0.298 (± 0.025)	0.303 (± 0.026)	0.291 (± 0.025)	0.296 (± 0.025)

Table 3.2 – Model run statistics and support for "successful" recovery in different biomes and under passive/active restoration. Columns indicate proportions of resample model runs supporting criteria of: best model %DE $>$ 10 (column 1), age coefficient positive (column 2), both %DE and age criteria (column 3), complete recovery within 100 years (mean +/-s.d.) under passive and active restoration in forest (columns 4 and 5, respectively) and non-forest (columns 6 and 7, respectively). Data for restoration success based on Appendix Tables A.8.4 to A.8.6. Benefit of active restoration significant for all indicators (Wilcoxon signed rank test on paired data, $p < 0.001$). NA = Not Applicable

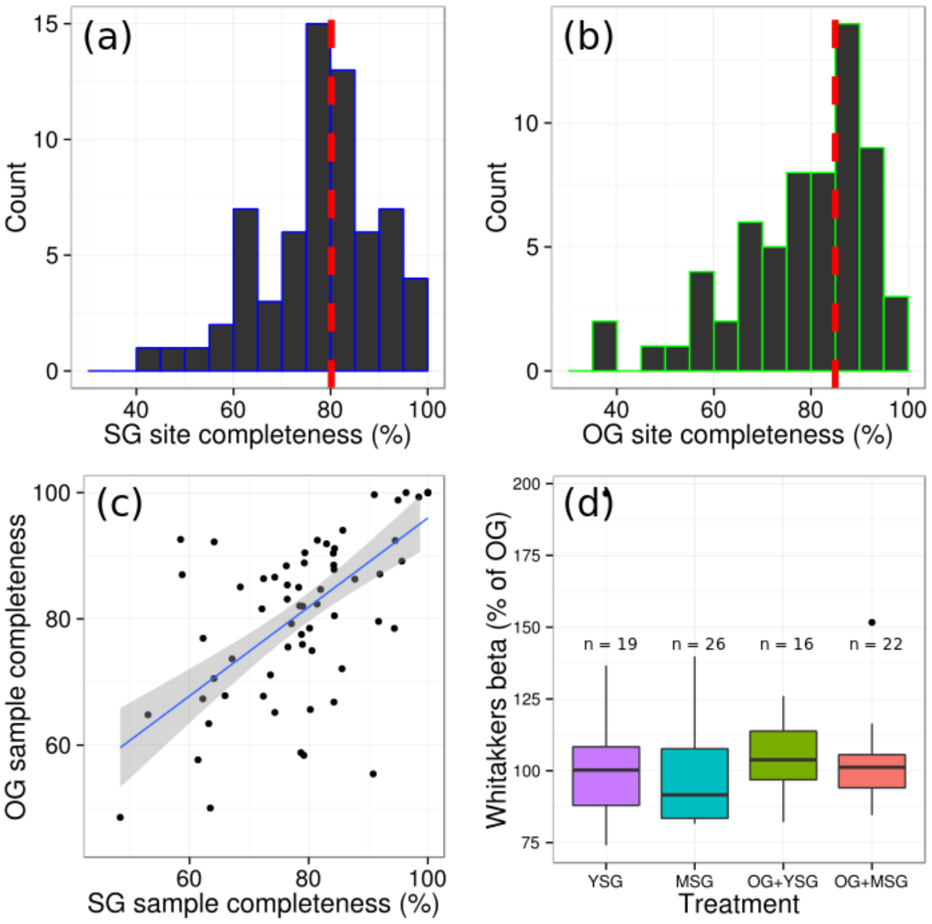

Figure 3.4 – Patterns in undersampling and turnover. Average study sample completeness across sites in secondary growth (a) and old growth (b) habitat, the distribution of paired differences in sample completeness (c), and Whittaker's beta calculated for young secondary habitat (YSG), mature secondary habitat (MSG) and pooled OG-SG data (OG+YSG, OG+MSG), expressed as percentage of average OG values (d). In general, differences in sample completeness between habitats (a, b, and c) were not statistically significant, and turnover patters in young (< 15 y) SG were as high as OG. However, middle-aged (> 15) SG exhibited a decrease in turnover relative to OG, indicating a more homogenous community, made up of a subset of OG species. Vertical dashed line in (a) and (b) indicate median values. Regression line in (c) represents robust linear regression with point-wise confidence intervals as shading. Box and whisker plots (d) show median values (central line), inner quartiles (box), outer quartiles (whiskers) and outliers (points).

not the case for mature SG). However, the observed differences in beta diversity between habitats were not significant (ANOVA; $n = 82$, $p = 0.587$).

3.4 Discussion

We analysed data from 108 comparative studies on the biodiversity value of passively recovering and actively restored habitat. We assessed whether there is empirical evidence to support biodiversity offset policy involving the use of habitat restoration as compensation for old growth habitat loss. To do this, we investigated hypotheses related to three criteria which must be met if offsets are to prevent a net-loss of biodiversity (Maron *et al.* , 2012): 1) restored ecosystems develop over time to harbour old growth assemblages of species, 2) active restoration significantly accelerates this process and 3) offset policy is designed and applied to accommodate the time-lags and uncertainties associated with compensatory habitat restoration. Our results support hypothesis 1 (convergence hypothesis) for all diversity indices except Fisher's alpha, demonstrating a significant trajectory towards OG values. When recovery does occur, we demonstrated that active restoration leads to significantly faster and more frequent recovery across a 100 y time horizon. However, even under active restoration, the combined uncertainties are large enough to require offset ratios that generally much higher that what is currently applied in practice.

Implications for restoration offset policy

Our study does not support the current form of implementation of restoration offsets to compensate the clearance of remnant old growth vegetation. Our models predict long time delays (decades to centuries) and a high probability of restoration failure (up to 81%), which is not accounted for by current offset policy. A number of previous studies on the success rate of offset and restoration projects supports our findings. In a recent synthesis of the offsetting literature, Bull *et al.* (2012) list (i) complete restoration failure, (ii) failure to persist, and (iii) development of novel communities as common grounds for the non-delivery of offset gains. These problems are mirrored in the restoration and secondary growth literature (e.g. Zedler & Callaway, 1999; Suding *et al.* , 2004; Munro *et al.* , 2009, 2011; Trimble & van Aarde, 2011; Woodcock *et al.* , 2011; Michael *et al.* , 2011). Lockwood and Pimm (1999) reviewed 87 active restoration projects finding only 6% that achieved a full recovery of community composition and structure. From a review of 240 studies, Jones and Schmitz (2009) found a success rate in recovery of species composition of ca. 23%. In a qualitative review, Suding (2011) estimated a overall success rate for replacing lost biodiversity value at less than 30%.

Ratio multipliers can be partly used to address the pitfalls mentioned above. Both information gap theory and time discounting can be used to calculate multipliers for uncertainty, restoration failure and time lags (Strange *et al.* , 2002; Carpenter *et al.* , 2007; Moilanen *et al.* , 2009a). However, our estimates of empirical success rates and time lags indicate that "robustly fair" offset ratios would be very large, easily exceeding what is commonly applied in practice. Based on existing issues of compliance in offsets, greatly increased equivalence ratios are likely to lead to an insurmountable institutional challenge, especially in socio-politically fragile regions with high levels of corruption. Even across existing offset schemes, Bull *et al.* (2012) found that insufficient compensation, partial implementation and policy change within the commitment period already represent common causes of offset failure and a net loss of habitat area, regardless of habitat quality (which we have investigated in this study). Larger offset ratios would also reduce the touted economic efficiency and flexibility of offsets and make them more politically challenging to implement and enforce f(Maron *et al.* , 2012; Moilanen *et al.* , 2009a; Bull *et al.* , 2012).

At least two caveats are worth mentioning in relation to these findings. First, our results apply to situations in which the loss of minimally disturbed, old growth vegetation is compensated. It is possible that in moderate or highly degraded areas, offsets are appropriate to prevent a net loss of biodiversity. However, offsets must compensate not only the current value of the impact site, but also the potential future value over the planning period (Moilanen *et al.* , 2009a), which may increase if degrading processes have subsided (leading again to higher multipliers). Additionally, many offset schemes currently concern the very type of remnant old growth habitat of conservation concern investigated in this study.

Second, we assumed that published restoration methods reflect current reality and are static over time. Our data included vegetation replanting using exotic species and low diversity seed mixtures. Improved techniques, such as staggered planting of a diverse set of native species has been shown to increase local species richness of birds and possibly other fauna (Munro *et al.* , 2011). However, the resulting habitat is more likely to be of secondary rather than core value to OG species (Munro *et al.* , 2011). Our data support this, in the *negative* relationship observed between Morisita-Horn similarity and SG patch size, which we interpret to indicate a spillover effect of individuals from close-by OG habitat driven by dispersal and temporary use rather than true occupancy (i.e. large SG patches are less likely to be influenced by edge effects, therefore show lower abundances of OG species and lower Morisita-Horn values).

Lessons for restoration ecology and conservation

Our results robustly support a strong positive effect of active restoration in “guiding” the process of biodiversity recovery (Suding, 2011). The stronger response of Sorenson similarity (60% improvement in recovery time) relative to Morisita-Horn (15% improvement) implies that species colonization occurs early, but convergence of assemblage structure lags behind. A significant improvement over all indicators supports the “foster ecosystem” hypothesis (Haggar *et al.* , 1997), which posits that restoration facilitates a rapid establishment of ecosystem structure and climate at the early stages of recovery through planting of fast growing vegetation (Figure 3.5). A favourable environment facilitates and accelerates the colonization and establishment of late-successional species, but still leads to very long recovery times for species relative-abundances (however, see

Figure 3.5 – The recovery of old-growth species assemblages in secondary-growth habitat is a long term process. (Left) Forest clearance and burning for agriculture leads to a rapid loss of compositional and structural diversity. (Middle) Within one or two decades, pioneer and mid-successional species establish relatively homogenous and depauperate stands. (Right) After several decades, a well-developed closed canopy, and associated environmental conditions (soil, climate and light conditions), facilitates the colonization of many old-growth species, but pre-disturbance species' relative abundance and resource use patterns take much longer to fully recover (centuries). (Left) Forest clearance in Afromontane forests in Northern Mozambique, (middle and right) 10– 15 yr and 30–40 yr secondary stands, respectively, in Central Kenya). Photo credits: M. Curran.

Munro *et al.* , 2009, who found no support for this theory in Southeast Australia). In any case, our results emphasize that ecological restoration should be employed as a complement to, rather than replacement for, conservation strategies based around the strict protection of old growth vegetation.

Sampling effort, success and spatial replication

Undersampling bias is a ubiquitous problem in diversity studies (Gotelli & Colwell, 2001). The results of this analysis confirm its presence across the secondary growth and restoration literature (Figure 3.4a-c). However, we found no systematic bias towards either SG or OG habitat, nor differences in beta-diversity between habitats (Gardner *et al.* , 2007a). Yet imbalances in sampling effort and the number of sites in the comparison showed moderately strong and important effects in the models. The GLMs predict that measured similarity will increase if either sampling effort is disproportionately invested in secondary growth habitat, or more sites are included in the comparison. These findings are consistent with Beck *et al.* (2013) who found that sample incompleteness inflates dissimilarities between samples, leading to overestimates of beta diversity. This implies that old growth species do not simply disappear altogether in SG habitats, but also become rarer. In all, we emphasize the need for more intensive surveys and replication at both the site and landscape scale in future studies (Chazdon *et al.* , 2009a; Bowen *et al.* , 2009; Munro *et al.* , 2011).

3.4.1 Conclusions

Our study has robustly assessed the ecological evidence base for restoration offsets and found little support that current theory and practice leads to a no net loss of biodiversity. Although we made many simplifications and assumptions in our analyses, we believe that the general trends and model parameters are robust, and our conclusions appropriate. In short, the complete recovery of old growth biodiversity is a very long and uncertain process that is significantly, but only partly, assisted by active habitat restoration. On the grounds of long time delays, high uncertainty and significant risk of failure, we conclude that a solid empirical foundation for restoration offsets to match the elaborate theory is currently lacking.

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Chapter 4

Pay the farmer, or buy the land?—Cost-effectiveness of payments for ecosystem services versus land purchases or easements in Central Kenya

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Abstract

Payments for Ecosystem/Environmental Services (PES) have recently emerged as an important tool in applied conservation. Yet evaluation of the economic and ecological benefits of PES over other *direct* conservation strategies remains limited. We prospectively compared the cost-effectiveness of PES to an alternative strategy of Land Purchases or Easements (LPE) in Central Kenya. We used Generalized Linear Models to spatially predict opportunity costs and land prices using household survey data supplemented with literature accounts. Conservation management and socio-economic development costs were sampled from four regional conservation organizations. These data were used to simulate a spatial selection process for conservation intervention (PES or LPE) based on ecological and economic criteria. This integrated simple land markets and rising agricultural productivity over time. Results indicate that, over a 30 year planning horizon, the LPE strategy led to larger and less fragmented reserves, representing 30% more species than PES. Non-discounted, cumulative costs per hectare were 40% lower for LPE due to high up-front and low operating costs. Regarding equity, our fairness criterion assumed the principle of compensation, but findings were similar when a flat-rate "egalitarian" yearly payment to fund development was included. Our results were strongly influenced by the discount rate (PES was favoured at rates >3%) and planning horizon (cost savings of LPE materialize only after 15–25 y). Wile we assumed transferable property rights and the institutional capacity to enforce compliance, Kenya's legal and policy framework restricts foreign land ownership and wider uptake of voluntary easements. Thus simpler institutional requirements make PES a more immediate and attractive property-based tool on the development frontier, despite lower cost-effectiveness in the long-term.

4.1 Introduction

The implementation of Payments for Ecosystem/Environmental Services (PES) to promote *in situ* biodiversity conservation on private land represents a relatively recent trend in applied conservation. This type of PES directly targets conservation values through activities such as the protection of habitat for endangered species, ecological restoration or the aversion of habitat loss (i.e. "conservation performance payments"; Ferraro & Kiss, 2002). PES represent an intuitive tool to periodically reward private landowners or communities, through annual cash or in-kind payments, for delivering additional conservation benefits (subject to a time-constrained agreement and strict conditionality). PES have been touted for their purported economic and ecological effectiveness, flexibility, and directness (Ferraro & Kiss, 2002; Wunder, 2005). However, research pertaining to their effectiveness in different contexts remains inadequate to draw generalized conclusions (Miteva *et al.* , 2012).

Existing literature that evaluates PES can be split into two broad classes: studies that assess the PES strategy in isolation, focusing on maximizing performance within a given context through design and implementation decisions (e.g. Pattanayak *et al.* , 2010; Wuenscher *et al.* , 2008; Wuenscher & Engel, 2012), and studies that compare the characteristics of PES to alternative conservation strategies (e.g. Ferraro & Simpson, 2002; Kiss, 2004). With regards to the latter *comparative* studies, there has been a tendency to qualitatively compare the *direct* PES approach to *indirect* conservation interventions (Integrated Conservation and Development Projects; ICDPs), that attempt to achieve conservation as a bi-product of environmentally-benign development.

Side-by-side quantitative comparisons of the PES approach to other, similarly *direct* conservation strategies are absent in the literature (Hanley *et al.* , 2012). An increasingly common direct and decentralized conservation strategy is simply to purchase land or land easements from private landowners (Armsworth & Sanchirico, 2008). The latter entails the (voluntary or involuntary) transfer of certain use rights from a landowner or *trustee,* who retains the title deed, to a *beneficiary.* While conservation land purchases or easements (hereby referred to as LPE) are more common in developed countries, they also operate extensively in the tropics, especially in Latin America and Sub-Saharan Africa (see Jones *et al.* , 2005; Armsworth & Sanchirico, 2008). Both strategies require a similar institutional framework, in terms of well-defined, secure and transferable property or exclusion rights, but important differences revolve around (Wunder, 2006):

- **Land market feedbacks and changing costs**: LPE may displace rural populations to urban areas, or elsewhere in the landscape, causing increased competition for urban services or alternative land parcels. This can cause market feedback affecting land prices and demand, potentially undermining initial conservation goals (e.g. Armsworth *et al.* , 2006). While PES do not lead to displacements, they do impact local land and labour markets (Wunder, 2006), are influenced by changing opportunity costs (e.g. Phelps *et al.* , 2013) and increase landowners' perceptions of the value of fallow or under-utilized land (Engel & Palmer, 2008; Engel *et al.* , 2008; García-Amado *et al.* , 2013).
- **Development and equity effects**: while LPE have no prior development aim, they can have strongly negative effects on equity and fairness in terms of landowner displacement and alienation of local communities (Fairhead *et al.* , 2012). While PES can have a similar effect (i.e. "green" land grabs for environmental purposes; Fairhead *et al.* , 2012), it is generally less extreme. PES scheme design has a strong influence on equity, and can result in the preservation, re-enforcement or reversal of prevailing inequalities in access to resources (Pascual *et al.* , 2010).

- **Property rights and enforceability**: LPE permanently transfer property/use rights, whereas PES only constrain use rights for a restricted period. The strength and self-enforceability of property rights also affects conservation costs, outcomes and therefore strategy choice. Weak governance (e.g. in frontier areas) may inhibit the enforcement of long-term easement agreements, making periodic monitoring and sanctioning a (costly) necessity (Wunder, 2005). Under weak property rights, the willingness to accept PES contracts may be negatively correlated to the ability to exclude outsiders, thus cost-effective conservation planning would allocate contracts to communities that cannot ensure the provisioning of a service (Engel & Palmer, 2008).

In this study, we present a comparative study of the cost-effectiveness of LPE versus PES in Central Kenya. Kenya is a diverse country encompassing a wide variety of climates and ecosystems, representing an appropriate spatial setting to investigate trade-offs and synergies between the two strategies across a gradient of agricultural suitability and livelihood strategies. The institutional framework for both PES and LPE is, in theory, present but underutilized under Kenyan law, and there are ongoing calls for wider uptake of both strategies (Rodriguez *et al.* , 2012; Gitahi & Fitzgerald, 2011; Norton-Griffiths, 1996). Our goal is to provide a first attempt at characterizing these two strategies in a joint ecological–economic model and to highlight the relevant factors influencing cost-effectiveness to guide further research (e.g. the temporal and spatial distribution of costs, discount rate, equity considerations).

In the next sections, we describe spatial economic models of net annual agricultural returns and land prices (Section 4.2.2), which are used to represent variations in the cost of PES and LPE respectively. We then present data on conservation management costs, collected from four regional conservation organizations (Section 4.2.3), and use these data to construct a spatial site selection algorithm for cost-effectively choosing new reserves. We investigate two equity scenarios of conservation intervention over a set time horizon with a fixed budget (Section 4.2.4), and compare outcomes with reference to efficiency–equity trade-offs and the policy framework in Kenya and other developing countries.

4.2 Methods

4.2.1 Study area

Our study region was located in Central Kenya, spanning 17 districts around the Aberdare Conservation Area and Mount Kenya (see study map, Appendix 8.3.2, Figure A.8.1).

The Aberdare Conservation Area is a 2'175 km^2 network of reserves consisting of the Aberdare National Park (766 km^2), the Aberdare Forest Reserve (1'033 km^2) and Kikuyu Escarpment Forest Reserve (376 km^2). It lies to the west of Mount Kenya, a World Heritage Site hosting the Mount Kenya National Park (715 km^2), Forest Reserve (2'010 km^2) and recently added Ngare Ndare forest reserve and migration corridor connecting montane populations of large wildlife with expansive northern rangeland habitats. For planning purposes, we divided the region in to 8 simple vegetation classes based on combinations of tree cover and elevation information (Appendix, Section 8.3.1).

4.2.2 Opportunity cost and land price models

Household surveys. Between March and May, 2012, we conducted 300 household surveys at two latitudinal zones adjacent to the Aberdare Conservation Area with the aim of estimating opportunity costs and land prices in the region (see Appendix 8.3.3 for sampling design and map of households). Our surveys gathered information from each household on (i) crop incomes and expenses for a single growing season, and (ii) income and expenditure data per month for livestock production and livestock product sales (e.g. milk, eggs, leather). We itemized input costs (labour, fertilizer etc.) and yields/off-take for all type of agricultural production (crops, livestock and livestock products) and aggregated them at the household level. Crop yields, livestock off-take and livestock products were multiplied by the stated selling price for that period to estimate gross household returns. For crop production, we converted gross seasonal returns to net annual returns by first calculating seasonal net returns (i.e. net returns = gross returns – gross costs), and then multiplying these by the number of growing seasons at that particular location, using a spatial dataset of the number of growing days (WRI, 2007) and an assumed growing season of 120 days (www.infonet-biovision.org). We included in our calculations all production, both consumed by the households and sold for cash income.

Modelling opportunity costs and land prices based on survey responses. We used the survey data, aggregated at the household level, to create separate spatial models for opportunity costs and stated land prices using Generalized Linear Models (GLMs). We corrected for spatial autocorrelation using the Generalized Least Squares (GLS) method, one of the best-performing existing methods (Beal *et al.* , 2010). The GLS method adopts the basic GLM formula:

$$y_i = \beta_0 + \sum_{h=1}^{p} \beta_h x_{hi} + \epsilon_i \tag{4.1}$$

where y_i is the ith response, x_{hi} is the ith value for the hth of p predictors, β_0 is the intercept,β_h is the correlation coefficient of the hth predictor and ϵ_i is an error term for the ith observation. Whereas in regular LMs, the error vector is assumed normally distributed around zero, $\epsilon_i \sim N(0, \sigma_\epsilon^2)$, the GLS approach adapts the error term to reflect observed pattern in spatial autocorrelation (correlation function) contained in the data, which may be non-normal. The correlation function was chosen by visually interpreting a semi-variogram (i.e. a plot of correlation values between points at different distances) and fit using penalized quasi-likelihood (Dormann *et al.* , 2007).

Both models used an identical model selection algorithm, which involved a range of local household-specific and spatial environmental variables as predictors (Appendix 8.3.3, Table A.8.2). We used an information-theoretic approach (Akaike's Information Criterion corrected for small samples; AICc) to select predictors and a multi-model averaging procedure to estimate coefficients (Burnham *et al.* , 2011). To investigate model fit and prediction error, we checked the structural goodness of fit of the best model (i.e. the "deviance explained", the proportion of null-model deviance that can be explained by the model) and also calculated total predicted model error using leave-one-out cross-validation (Appendix 8.3.3).

To spatially predict opportunity costs and land prices, we extracted predictor data using a 1 km point grid covering the entire study region. Response values were predicted for these points using the parameters of the best performing models for both opportunity costs and land prices. The resulting predictions were interpolated across the landscape using a bilinear spline with Tykhonov regularization (Brovelli *et al.* , 2004), choosing a spline neighbourhood size of 10 km (the radius of points which individual splines are fitted to the data) and regularization parameter of 0.4 (allowing moderate deviation of the spline from the interpolated data points).

Integrating literature opportunity cost data. To improve our spatial model of opportunity costs, especially for lower arid regions and production systems where we lacked data, we searched for peer-reviewed and grey literature documenting net revenues from agriculture, woodlots, and forest plantations (stumpage fees), and classified these according to the Food and Agriculture Organization's (FAO) seven agroecological zones of Kenya (www.fao.org) using spatial data from WRI (2007). We made necessary inflation adjustments, and spatially mapped these values by attributing them to a land cover classes resulting from a cross between a simplified version of the Africover land cover product (Africover, 2004) and the seven Agroecological Zones of Kenya (1: per-humid, 2: humid, 3: semi-humid, 4: transitional, 5: semi-arid, 6: arid and 7: per-arid). We weighed values by the number of growing seasons, and combined the resulting map with our empirical model (described above) by averaging predicted opportunity costs across the two maps. A full description of this procedure is contained in Appendix 8.3.3. We used this consensus OC map in our scenarios because it better represented Agroecological Zones and agricultural strategies in the study region than either model alone.

Accounting for productivity increases and land market feedbacks. For our scenario analysis (see below, Section 4.2.4), we made two changes to the baseline opportunity cost and land price models. To simulate increasing agricultural productivity and/or higher prices due to future scarcity of land or other inputs, we developed a model of high opportunity costs, which equalled a 50% increase on current values, and reduced the deduction for crop failure by 50% (i.e. land use patterns change to more profitable uses, farming practices are more efficient and successful). In reality, the price effect could be reduced through imports, but future agricultural intensification will likely increase opportunity costs non-the-less (e.g. Phelps *et al.* , 2013). For land prices, market feedbacks are likely to be more severe, as land is immobile and non-substitutable (i.e. land cannot be "imported" to satisfy rising demand). Land market feedbacks have previously been shown to negatively affect the effectiveness of land purchases for conservation (Armsworth *et al.* , 2006). In our scenarios, we simulated a simple land market to integrate demand-side feedbacks on land prices as increasing amounts of land are purchased and set aside for conservation. We based this model on the log-linear relationship between the amount of land developed within a 21 x 21 km neighbourhood, and the average predicted land price from our model (similar to Armsworth *et al.* , 2006), with further information provided in Appendix 8.3.3.

4.2.3 Conservation management costs

Cost categories. In May 2012, we conducted semi-structured interviews with representatives of four conservation organizations in the study area in order to collect data on conservation management costs, which were classified as follows. We distinguished between investment (start-up) and operational (running) phases, and subdivided these into transaction and production costs according to Mburu *et al.* (2003), itemizing broad categories of both cost types for each organization. *Production costs* encompass the costs of infrastructure, salaries, equipment, training etc. required to deliver conservation outcomes. Additional expenditures required to prevent or compensate damages caused by wildlife/biodiversity, and the costs of foregone resource use (opportunity costs), also fall under production costs if they are realized through a compensation scheme. In contrast, *transaction costs* encompass search and information costs, bargaining, decision-making and the monitoring and enforcement of compliance. A detailed description of the items in each cost category is contained in the Appendix 8.3.4.

Phase and cost category	**Conservation strategy**	
	Land purchase or easements	**Payments for ecosystem services**
Investment phase		
Transaction costs	finding sites, negotiating purchase	finding sites, negotiating contracts
Production costs	land value and NGO processing fee on purchase [X]; habitat rehabilitation; equipment; infrastructure, material and labour to build headquarters	land value and NGO processing fee for demonstration site [X]; habitat rehabilitation; equipment; infrastructure, material and labour to build headquarters
Operational phase		
Transaction costs	periodic negotiations ("peace meetings") with adjacent landowners [X]	regular meetings with PES providers [X]; monitoring and negotiations [X]
Production costs	material and labour for operations (strict protection); administration [X]; capital maintenance	material and labour for operations (strict protection); administration [X]; capital maintenance; opportunity costs of PES providers [X]

Table 4.1 – Cost items included in the 30 year scenarios for the two conservation strategies. Items that differed across strategies are marked with "[X]", and occurred mainly in the operational phase. None of the items for the "development" scenario differed between strategies, and therefore were omitted from the table.

Conservation organizations. The four organizations interviewed were the Lewa Conservancy (http://www.lewa.org), Il Ngwesi Conservancy (http://ilngwesi.com/), Mount Kenya Biodiversity and Conservation Group (MtKeBio) and Friends of Kinangop Plateau (FoKP), with detailed profiles of each provided in Appendix 8.3.4. The organizations were chosen to represent conservation interventions in two dominant ecological zones in the study region: Arid and semi-arid rangelands traditionally used for extensive ranching by pastoral communities (Lewa and Il Ngwesi), and montane habitats in the buffer zones of the Aberdares Conservation Area and Mount Kenya, consisting of evergreen montane forest (MtKeBio) and montane tussock grassland (FoKP), traditionally cultivated for subsistence crops, cash crops (e.g. tea and coffee) or used for pasturing.

Strategy-specific costs. In contrast to the LPE strategy, PES are not yet being used by the organizations interviewed, thus some of our transaction cost estimates were derived through a mix of scenario building, and transferring costs from similar activities (e.g. targeting and attracting member households, or negotiation and monitoring ICDP activities). More details are provided in Appendix 8.3.4.

4.2.4 Scenario development and strategy comparison

To investigate the cost-effectiveness of the two conservation strategies (PES and LPE), we developed two scenarios, calibrated using the data described above. We adopted the perspective of a single conservation organization with a fixed investment and operational budget. (derived through extrapolating budgets of the interviewed organizations to a total of 10 montane and 5 rangeland organizations, a working estimate of the number of established BirdLife Site Support Groups and rangeland reserves in the region). We developed a targeting algorithm to cycle through each vegetation class and set aside land for conservation (via PES or LPE) of high ecological value and low financial cost. We used a time horizon of 30 years, which represents a very low estimate of the time necessary for the species composition of forest and rangeland assemblages to recover to "old growth" values under a regime of active restoration (Curran *et al.* , 2014).

Equity scenarios. Within the strategy choice of LPE or PES, we applied two "fairness criteria" to our scenarios, following the classification of Pascual *et al.* (2010). In the baseline scenario, our models followed a "compensation" criterion, where payments were differentiated based on the costs of participation (based on spatial patterns of production and transaction costs). A second equity scenario modified the fairness criterion to incorporate an *additional* "egalitarian" development payment on top of the compensation payments. This was scaled according to expenditure data for infrastructure (e.g. road improvements, housing), education (e.g. scholarships, school financing, adult literacy programmes), sanitation (e.g. waste management, water harvesting and treatment) and health care (e.g. medical centres, reproductive health education) from existing development programmes implemented by the interviewed organizations. Regarding the two strategies, the "development" payments could be thought of as a direct cash or in-kind payment to participants of a PES scheme above compensation requirements, or payments that are dispersed to nearby communities under an LPE scheme to address alienation and exclusion. We refer to these scenarios in the text simply as "baseline" and "development". Table 4.1 describes the cost and benefit items included in the baseline conservation scenario for each strategy ("development" items are not shown as they did not differ between strategies).

Site selection. For the 30 year scenarios, we assumed that all land outside public protected areas was unprotected, and developed a targeting system aiming for 25% conservation are coverage of each vegetation class. Within each class, criteria were set to target areas of lowest economic cost (either land prices or opportunity costs, depending on the strategy) subject to the following constraints: pixels were chosen of higher-than-average *mammal species diversity* (using distribution data from a recent IUCN

Global Mammal Assessment; Rondinini *et al.* , 2011a), higher-than-average *proximity to a public protected area*s, and lower-then-average *development intensity* (based on the proportion of a 1 km cell occupied by a "developed" land cover class—agriculture, pasture, infrastructure, urban, or plantation). Land prices and opportunity costs increased after each time step, in response to our simulated land markets and productivity gains. For both scenarios, establishing a new conservation area involved investing in habitat restoration, capacity building, infrastructure, equipment etc. (Tab. 4.1). A more complete description of the ecological benefit functions and site-selection algorithm are provided in Appendix 8.3.5. Importantly, our scenarios apply present costs to historical change (i.e. we reconstructed the current private conservation network based on patterns of current prices and opportunity costs). They should not therefore be compared to actual existing patterns of reserves in the region, which developed under very different historical circumstances.

We compared PES and LPE strategies in terms of conservation outcomes and cost effectiveness. To quantify conservation effectiveness, we used three indicators of biodiversity benefit, quantified at the end of the modelling period (30 y): (i) the area of new land secured under conservation management, excluding the existing PA network (in km^2); (ii) the ratio of boundary length to reserve area (BL:RA) of the new reserve network in the region, used as an indicator of spatial agglomeration with higher values indicating more fragmented reserves; (ii) the proportion of mammals species in the region represented by the reserve network, referred to as "conservation coverage*"*, estimated using a simple Arrhenius (1921) variant of the species area relationship (see Appendix 8.3.5 for details). To quantify economic effectiveness, we calculated cumulative conservation costs over the entire modelling period, and expressed this per unit area (km^2) and per percentage gain in conservation coverage ($\$/C_{gain}$). We also aggregated these data for each of the 8 vegetation classes. To see how the choice of conservation strategy is influenced by the nature of investment decisions and the time profile of costs, we applied a range of discount rates to future cash flows (1–5%), assuming a public policy and social cost-benefit analysis (CBA) perspective when choosing the range of discount rates (see the discussion for more details).

Geographic data manipulation, analysis and visualization were performed using GRASS GIS (vers. 6.4.3) and Quantum GIS (vers. 2.0). All statistical analyses were performed with the statistical programming language R (vers. 2.15.3), particularly the packages "glmulti" (model selection and averaging), "nlme" (GLS), "spgrass6" and "rgdal" (spatial functions). Data preparation and some summary statistics were calculated using PostgreSQL (vers. 9.2) and LibreOffice Calc (vers. 3.6.2.2).

4.3 Results

4.3.1 Mapping spatial conservation costs

Model selection. The model selection procedure led to four predictors in the final opportunity cost model (percentage tree cover, aspect heterogeneity, mean diurnal temperature range, distance to nearest water body and distance to nearest road or minor town, and slope) and 6 variables in the land price model (aspect, mean diurnal temperature range, minimum temperature of the coldest month, distance to nearest minor town, and percentage tree cover). For opportunity costs, the structural goodness of fit of the best GLM from the model selection procedure was 18.2%, indicating a substantial amount of unexplained variation. Leave-one-out cross-validation also revealed an average prediction errors of \$887 (about a third of the median sample value of \$2'148.9). Goodness of fit for the land price model was better, with a higher explanatory power than the agricultural returns model (47.4% deviance explained), but leave-one-out cross-validation still indicated a substantial average prediction error (\$5'520 or about a third of the median sample value of \$14'880). Coefficients for each predictor are provided in the Appendices (Table A.8.3).

Literature data. We found nine publications with information on opportunity costs, with similar value ranges to our modelled data (Appendix 4.1, Table A.8.4). We spatially mapped of these literature values (see Methods 4.2.2) and combined them with our predicted opportunity costs. Final opportunity cost and land price models are shown in Figure 4.1. In general, opportunity costs were predicted to be highest (max = \$16'269 $ha^{-1}y^{-1}$) in the mid altitudes and around Nairobi and major infrastructure networks (Figure 4.1a). Patterns in land prices resembled opportunity costs to some degree (reflecting capitalized agricultural returns), but showed no link to transport networks, and a stronger link to Nairobi and its surroundings (max = \$29'903 ha^{-1}; Fig. 4.1, right).

Management costs. Our management cost data illustrated variable costs across organizations, and is presented fully in Appendix 8.3.4 (per-hectare values applied to our spatial models are presented in Table A.8.6). In general, transaction costs were lower in rangelands than montane areas, particularly for the PES strategy (e.g. PES transaction costs of \$1.5 ha^{-1}, \$14.3 ha^{-1} and \$62 ha^{-1}, for rangeland, montane grassland and forest areas respectively). This can be attributed to a higher population density and thus more providers per area conserved. In general, more investment was required for forest ecosystems due high estimated forest rehabilitation costs (\$3'965 ha^{-1} for sparse/degraded forest) and infrastructure costs during reserve creation (\$384/ha).

4.3.2 Organization management costs

4.3.3 Scenario outcomes

For the LPE scenarios, we used a total investment budget of $1'322 million, and an annual budget of $67.1 million thereafter (an extrapolation of the combined optimal budgets of the four interviewed organizations to the broader landscape; see Appendix 8.3.4). The PES scenario was identical in terms of total funds, but with a smaller investment budget, and the remainder spread over the 30 year horizon as additional operational funds.

Overall, the scenario analyses revealed LPE to be more effective in securing conservation gains throughout the 30 year period, largely due to high up-front but low operating costs. Total reserve area at the end of the period was 3'971 km^2 for LPE versus 2'618 km^2 for PES, a factor difference of roughly 1.5 (Figures 4.1c, 4.1d, and 4.2a). Differences were lower for the "development" scenario (2'330 km^2 vs. 1'898 km^2; factor of 1.23). In terms of time profiles, both strategies had similar reserve sizes after the investment phase (the PES reserve was slightly larger with 2'322 km^2 compared to 2'242 km^2 for LPE). Within about a decade the value of total PES contracts equalled the operational budget, and growth in conservation area plateaued. In contrast, the LPE strategy steadily reinvested a declining proportion of the operational budget into new reserve purchases.

Patterns in conservation effectiveness (i.e. fragmentation and species representation) generally reinforced the advantage of LPE (Table 4.2). The LPE strategy led to a larger average habitat patch size (by a mean factor of 2.03 across scenarios, compared to PES) and lower perimeter:area ratio (LPE value 16% lower than PES). Total increase in conservation coverage, expressed as the percentage increase in species representation predicted by the SAR, indicated that about 30% more species were represented by the LPE strategy.

Economically, the LPE strategy was 23% more cost-effective with no discounting (measured in cumulative costs per percent increase in conservation coverage). However, at a discount rate of 3% the strategies did not differ greatly, and at 5% the PES strategy showed a clear cost-advantage at the end of the modelling period (both in terms of costs per unit conservation coverage and unit area; Table 4.2 and the Appendices, Figure A.8.4). Finally, it is worth noting that the cost-effectiveness of the LPE strategy was dependent on the planning horizon, as cumulative savings only offset the up-front costs after ca. 15–25 years (range observed based on both "baseline" and "development" scenarios discounting up to 1%). For projects with a lower planning horizon, results would have favoured PES.

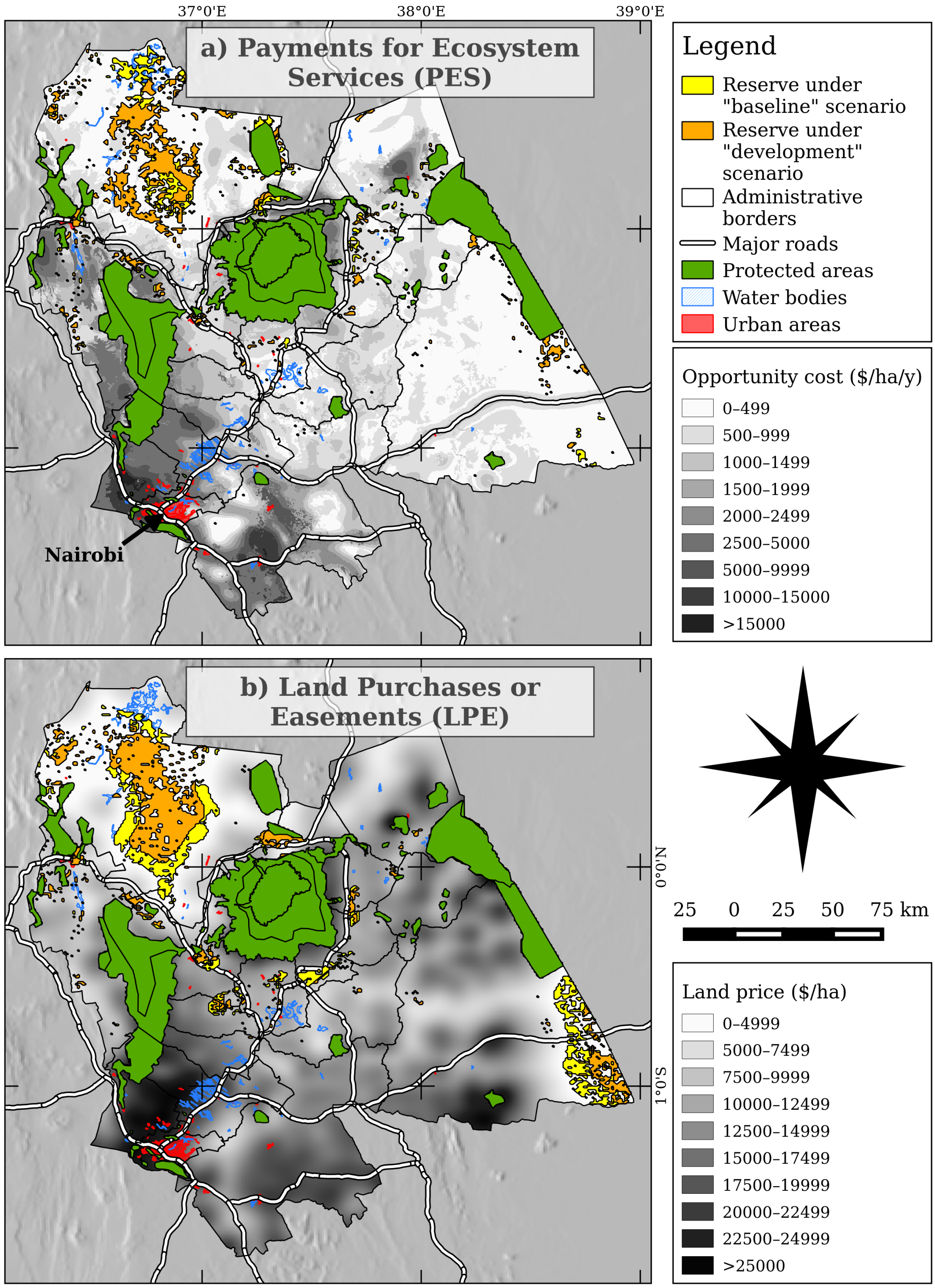

Figure 4.1 – Scenario and model results showing conservation coverage after 30 y using PES (a) and LPE (b) strategies, for both "baseline" and "development" scenarios (see figure legend). Underlying layers show final opportunity cost (a) and land price (b) models for PES and LPE, respectively. The "development" scenario reserves represent a subset of the baseline scenario, constrained by higher operating costs but otherwise using an identical targeting scheme.

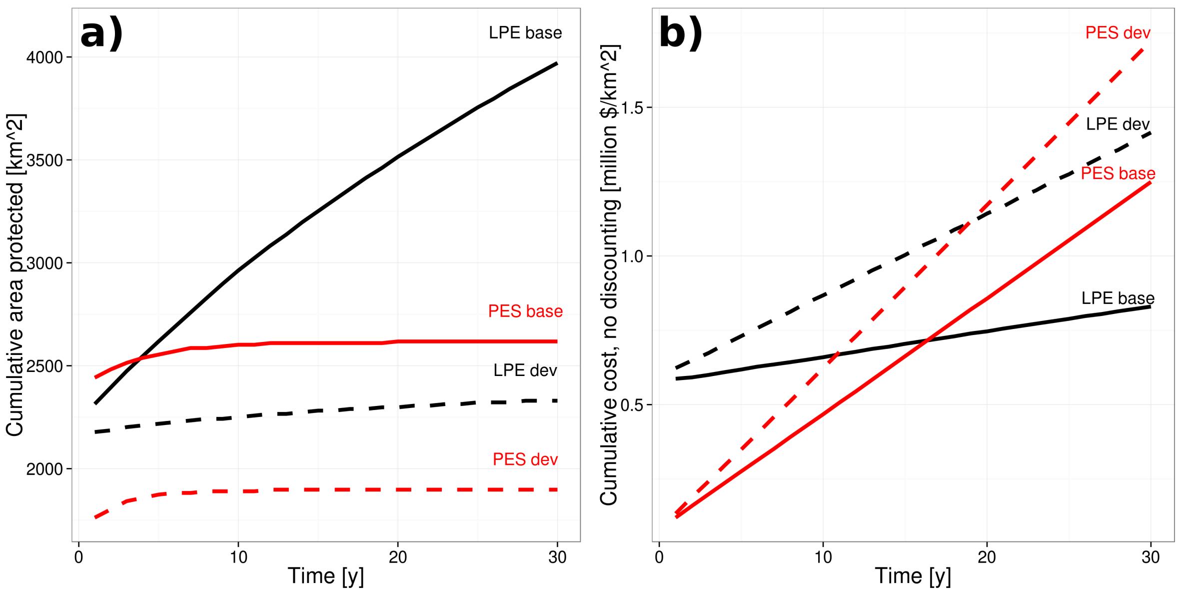

Figure 4.2 – Scenario results over a 30 year time horizon. Illustrates changes in total conservation coverage in km^2 (a) and cumulative costs per unit area (b). Non-discounted cumulative costs per hectare always increase over time, as future interventions are more costly due to diminishing returns (i.e. the cheapest contracts or land parcels are purchased first).

Indicator	**Baseline**		**Development**		**Ave. ratio**
	LPE	**PES**	**LPE**	**PES**	**LPE:PES**
Mean patch area [km^2]	11.27	5.35	8.28	4.22	2.03
Mean patch perimeter [km]	9.42	7.64	7.89	6.83	1.19
Perimeter:area ratio	0.84	1.43	0.95	1.62	0.59
Cons. coverage gain [%], region	6.9	4.8	4.3	3.6	1.32
Cons. coverage gain [%], Veg1	12.2	8.6	7.8	6.7	1.29
Cons. coverage gain [%], Veg2	13.8	13.6	12.4	10.3	1.11
Cons. coverage gain [%], Veg3	17.0	11.3	10.4	8.5	1.36
Cons. coverage gain [%], Veg4	0.9	0.8	0.9	0.8	1.04
Average costs [$/% cov. gain]	478	684	766	916	0.77
Average costs [$/% cov. gain] @ 1%	415	544	664	725	0.84
Average costs [$/% cov. gain] @ 3%	320	335	513	439	1.06
Average costs [$/% cov. gain] @ 5%	268	218	428	280	1.38

Table 4.2 – Indicators of conservation cost-effectiveness for the two strategies (LPE and PES) across the two scenarios (baseline, development). Average costs calculated per percentage increase in conservation coverage over the scenarios. Coverage is expressed in terms of the proportion of species regionally protected by the reserve system, predicted by the species area relationship. The final column ("Ave. ratio") expresses the relative performance of LPE over PES. Result shown for three discount rates (1%, 3%, 5%), and separately for each vegetation community (1–4; communities 5–8 not shown because they were adequately protected under public PAs). Veg1 = Low-elevation grassland and bush, Veg2 = High-elevation grassland and bush, Veg3 = Low-elevation sparse or degraded forest, Veg5 = Mid-elevation sparse or degraded forest.

4.4 Discussion

In this study, we quantitatively compared the cost-effectiveness of PES with a strategy of land purchases or easements (LPE) in Central Kenya using spatial ecological-economic models. Our results suggest that the LPE strategy results in larger and less fragmented reserves, representing 30% more species than PES. Cumulative costs per unit conservation coverage were lower, but this was dependent on discount rate (PES was favoured >3%) and planning horizon (LPE exhibited an advantage in cumulative non-discounted costs at 15–25 y). While the spatial determinants of costs in our models played a role (e.g. land price hotspots around population centres, biophysical determinants of opportunity costs) they only translated into subtle differences in reserve placement between strategies (Figure 4.1). Differences in the temporal allocation of costs (i.e. investment vs. operational costs) had a predictably profound effect. One key consideration is the degree to which agricultural returns are "capitalized" into land values (Naidoo & Adamowicz, 2006). In other words, the ratio of agricultural returns to land prices, which partly determines the temporal break-even point where one strategy is favoured over another (in our case, 15–25 y). This will be context-specific, influenced by socio-economic (e.g. technology, poverty, individual discount rate), cultural (e.g. non-monetary land values) and institutional (e.g. land tenure security, legal system) factors (Holden *et al.* , 1998). Our household survey data indicated a median value of 0.144 (calculated as ratio of OC to LP values in Table A.8.1 from Appendix 8.3.2). Higher documented rates in other developing regions (e.g. Naidoo & Adamowicz, 2006; Holden *et al.* , 1998) suggest our results could be transferable, since higher rats favour the LPE strategy.

Choice of discount rate and cost-effectiveness

We used cumulative net present conservation costs ($/ha) to assess cost-effectiveness over the project planning horizon. We adopted a public policy perspective, applying a range of *social* discount rates from 0–5%. In environmental cost benefit analysis (CBA), the choice of appropriate discount rate is intensely debated by economists (Gowdy *et al.* , 2011; Atkinson & Mourato, 2008). Some authors recommend using the market rate of interest, i.e. the rate of return on investments, corrected for inflation, investment risk and population growth (Lind, 1982; Nordhaus, 2007). This indicates how fast per-capita wealth is increasing, with the concurrent assumption that a richer future can afford higher costs than the present. In this context, our rates appear low in comparison to real interest rates in Kenya, which averaged 7.45% over the past decade (World Bank, 2013b). However, correcting for population growth in the same period (2.7%) gives a lower rate of 4.75%. Adjusting for risk can have a strong effect, as rates for "safe" and "risky" investments differ substantially (e.g. in the U.S. since 1920, "safe" short term government bonds yielded on average 1% versus 7% for "risky" corporate stocks; Cochrane, 2001). Ideally, policies with wide-reaching and potentially irreversible consequences (such as biodiversity loss; Gowdy *et al.* , 2011) should receive a conservative, risk-averse rate. This is the case for economic assessments of climate change, which generally apply low values ranging 1.5–2% (Quiggin, 2008).

A second line of argumentation rejects the use of interest rates (or other proxies of economic growth) altogether due of the implicit assumption that the future will be better off than the present. This has been strongly challenged on the grounds that apparent productivity and wealth increases in the recent past are partly driven by unsustainable consumption (drawdown) of natural capital and loss of ecosystem services (e.g. Gowdy, 2007; De Groot *et al.* , 2013; Gowdy *et al.* , 2011). Projecting such rates into the future in the form of expected returns (interest rates) paints an overly optimistic picture, justifying a high level of discounting and temporal cost-shifting (known as the "optimists paradox"; TEEB, 2008). If environmental costs are taken into account, resulting rates can be low or even negative. For example, Xepapadeas & Vouvaki (2009) investigated productivity growth in energy production across 23 countries, concluding that when the estimated external costs of CO2 emissions are included, the average growth rate turns negative (implying a poorer rather than richer future, and a negative discount rate). In addition, many environmental costs are irreversible (e.g. species extinction), and increasing scarcity of non-substitutable ecosystem goods and services means their value is likely to increase in the future, re-enforcing the argument to discount environmental benefits (or the costs of environmental protection) at a low or negative rate (De Groot *et al.* , 2013).

With regards to Kenya, the World Bank (2013b) estimates that the environmental costs of carbon emissions, mineral depletion, forest loss and other types of natural resource depletion incurred an average annual cost of 2.8% of GDP over the past decade (compare this to GDP growth of 4.25% per year for the same period). Biophysical valuation using embodied energy analysis has given rise to an annual figure of 3.8% of GDP for the costs of soil erosion alone (Cohen *et al.* , 2006). Considering the above, there is simply no *a priori* reason to assume a high, or even positive, discount rate when assessing cost-effectiveness of conservation policy in Kenya. We therefore base our conclusions on results using a low rate, which favours the cost advantage of the LPE strategy (i.e. at rates below 3%, costs are about 10% lower than PES).

Equity–efficiency interdependency and institutional regimes

Development dimension and equity. Criticisms of the LPE strategy include the lack of an integrated development dimension and risk of displacing local stakeholders (Wunder, 2006). PES can also have strongly negative social effects by introducing a new source of income and changing access/use rights to resources. Both strategies may lead to conflicts between social groups, especially in regions where informal (customary) arrangements or powerful interests prevail (Fairhead *et al.* , 2012). Cost-effectiveness should thus only be assessed within an explicit equity framework, as cost-effective solutions may suffer from a lack of legitimacy when participation and benefits are biased (Swallow *et al.* , 2009). Pascual *et al.* (2010) propose a set of "fairness criteria" to which PES or other interventions align. These range from a needs-based "maxi-min" criterion at one extreme (i.e. paying the maximum amount to the poorest providers to explicitly counteract existing inequalities) to a "status quo" criterion at the other extreme (i.e. scaling payments to match income patterns, thus re-enforcing prevailing inequalities).

Our "baseline" scenarios for both PES and LPE assumed a "compensation" fairness criterion, where expenditures (PES or LPE) are differentiated based on estimated costs (i.e. opportunity costs and land prices). Under this fairness criterion, participation favours the poor ("pro-poor PES") if wealth is related to costs (e.g. small, subsistence land owners accept lower payments whereas larger farms have higher opportunity costs due to industrialization and scale). However, our household survey data revealed a weak, but significant, *negative* correlation between log-transformed farm size and both ranked opportunity costs ($n = 227$, $p = <0.001$, $R^2 = 0.092$) and ranked land prices ($n = 227$, $p = 0.002$, $R^2 = 0.043$). Thus, a cost-effective PES or LPE strategy would favour larger land holders (which would also reduce transaction costs), re-enforcing existing wealth inequalities unless a more distributive fairness criterion were applied.

To investigate an alternative equity setting, we included a "development" scenario, which added a fixed development premium. In terms of equity model, this represents an additional "egalitarian" payment (i.e. fixed payment per ha for all participants; Pascual *et al.* , 2010) on top of the compensation payment. Comparing across scenarios shows that LPE with "development" predicted to be only 10% more expensive than the "baseline" PES scenario, with no development effect (Figure 4.2 and Table 4.2). This illustrates that a prior development goal *could* be integrated into a fully direct strategy such as LPE, resulting in only moderate increases in overall costs compared to a PES baseline. This raises the question of whether a landowner or community is better off under subsistence PES payments, but with intact tenure and use rights, or with a once-off payment in the form of an easement in perpetuity, but with continued development assistance? Answering these questions lie beyond the scope of our study, but highlight the importance of considering both equity and efficiency criteria simultaneously when comparing conservation strategies (i.e. for the price of strategy X with no equity effect, could a better equity outcome result from strategy Y?).

Property rights and governance. Our analysis assumed transferable and enforceable property rights, which may be considered unrealistic both in Kenya and elsewhere on the development frontier. In Kenya, legislation limits ownership of agricultural freehold land to Kenyan nationals, restricting options for purchases by non-Kenyan organizations (this also applies to the shareholders of land-owning companies or beneficiaries of a land trust; Laws Laws of Kenya, 2010). Purchases must therefore be made through organizations registered in Kenya, who retain title deeds (e.g. the Birdlife International "site support groups interviewed in this study utilize donor funds in this manner). There are no restrictions on the ownership of easements, but the Kenyan legal framework only facilitates *involuntarily* easements obtained through a court order (Laws Laws of Kenya, 1999). The law has also never been invoked in practice (Gitahi & Fitzgerald, 2011; Kipkoech *et al.* , 2011). Voluntary leasing contracts between a landowner/community and a third party (e.g. NRT community land trusts) are similar to easements in terms of legal status, but are not in perpetuity and cannot be sold or inherited. Therefore the institutional framework for wider uptake of voluntary easements outside of the courts between private parties is currently lacking (the "Kenya Land Conservation Trust" is actively lobbying to address this; Gitahi & Fitzgerald, 2011).

Easements and PES assume landowners have the capacity to exclude outsiders, and that governance capacity is strong enough to enforce agreements and penalize non-compliance. While our scenarios included proxies of exclusion costs (fence construction, guards etc.) and monitoring (e.g. site visits, stakeholder meetings) for both strategies, we did not include any costs of non-compliance. In Kenya, the capacity to enforce environmental legislation is weak (Gitahi & Fitzgerald, 2011). Risks of non-compliance, and the associated cost of sanctioning (e.g. through a judicial process), could differ between strategies. In such a context, short-term, regular PES payments intuitively represent a more practical means of providing an incentive for compliance (Wunder, 2006). PES are also facilitated by a number of existing legal statues in Kenya (Kipkoech *et al.* , 2011). Conservation leasing, mentioned above, represent a form of PES employed by some conservation entities to maintain land for wildlife migration between National Parks and other areas (Gitahi & Fitzgerald, 2011; Rodriguez *et al.* , 2012; Norton-Griffiths, 1996). In this respect, PES have an advantage for immediate deployment. Finally, if the time horizon is modest (i.e. <15–25 y), PES would prove to be the more cost-effective option.

4.5 Conclusions

Assessing the cost-effectiveness of competing conservation strategies requires the consolidation of spatially and temporally distributed costs across an appropriate project planning horizon. This raises tricky methodological issues and uncertainties, particularly in spatially mapping costs that are both socially and biophysically determined, predicting their future trajectory and choosing an appropriate discount rate. Our research suggests ways to address these issues when comparing the cost-effectiveness of payments for ecosystem services (PES) with land purchases or easements (LPE). Assuming a low discount rate, our results suggest a substantial advantage of LPE in the long term, but also illustrate a window of opportunity for effective deployment of PES in the short-to-medium term, which is enhanced by simpler institutional requirements. Thus future research should prioritize assessing these and other strategies in combination, integrating more diverse equity scenarios, explicitly measuring the likely equity of these scenarios (e.g. regional changes in the Gini index) and better represent important criteria such as additionality, leakage and permanence.

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Chapter 5

High resolution assessment of land use impacts on biodiversity in Life Cycle Assessment using habitat suitability models

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Abstract

Global agricultural production and trade contribute massively to biodiversity loss. Life cycle assessment (LCA) studies help inform decision makers about direct and displaced environmental impacts of products and are thus important to the agricultural intensification debate within conservation. However, largely due to poor communication between disciplines and underdeveloped LCA methods, the LCA perspective is rarely adopted. To address this issue, we present a novel LCA method based on the increased extinction risk of mammal species to model product-related impacts of land use on biodiversity. We contrast it with two recently developed LCA methods (focusing on local and regional impacts, respectively) and illustrate the application of these methods to major export crops from East Africa (tea, coffee, and tobacco). The methods highlight hotspots of product-related biodiversity impacts. We encourage conservationists to further apply and develop such models to improve our understanding of the link between production, consumption, and biodiversity loss.

5.1 Introduction

Agricultural land use is a main driver of current and expected future global biodiversity loss (Sala *et al.* , 2000; Visconti *et al.* , 2011). In many developing countries, large areas rich in biodiversity have been converted to cash-crop production for developed countries. Lenzen *et al.* (2012) estimated that international trade is responsible for about 30% of global threats to species. Consumers and policy makers are often unaware of these implications, and informing them about the displaced biodiversity impacts of products is a first step toward more sustainable consumption patterns and reducing rates of global biodiversity loss.

Life cycle assessment (LCA), which quantifies the impacts of products over their entire life cycle (ISO, 2006), is a well-established methodology to capture displaced environmental impacts. It helps to identify environmental hotspots in internationally distributed value chains of products and can highlight potential trade-offs between different environmental compartments (e.g., reducing a product's contribution to climate change while increasing the pressures on biodiversity). In this sense, the life cycle perspective is extremely important to the agricultural intensification debate within conservation (i.e. "land sparing" vs. "land sharing"; Tscharntke *et al.* , 2012a) because it helps to illuminate the "hidden costs" of intensive agriculture (Vandermeer *et al.* , 2005) driven by indirect inputs (e.g., the land used to mine phosphorus and grow fodder or the ecotoxic impacts of pesticides). LCA thus helps to illuminate the complex chain of causalities involved in what may appear to be local threats to conservation. This is extremely important in the context of global commitments towards biodiversity conservation, i.e. the Convention on Biological Diversity Aichi targets (CBD, 2013). Target 4 aims at the implementation of plans for sustainable production and consumption by governments, business, and stakeholders and target 2 at the integration of biodiversity values into national accounting. Here, LCA can play an important role as it can directly quantify the extent to which changes in consumption and production pattern affect land use change and biodiversity loss. Yet conservation scientists rarely adopt a life cycle perspective, which we attribute both to a lack of intradisciplinary exchange and of adequate methodologies to assess biodiversity loss in LCA (Curran *et al.* , 2011).

Methods implemented in standard LCA software only assess the impacts of agricultural land use on biodiversity in Europe (De Schryver *et al.* , 2010; Koellner & Scholz, 2008). Two globally applicable and operational LCA methods for assessing the impacts of land use on biodiversity were developed recently (de Baan *et al.* , 2013b,a). The first method assesses the relative local reduction of species richness (*R-Local*) on different types of human-modified land (e.g., annual and permanent crops) compared to undisturbed areas (de Baan *et al.* , 2013b). It is based on data derived from a global literature review (GLOBIO3; Alkemade *et al.* , 2009) and national biodiversity monitoring data (BDM, 2004). For each biome, data points are grouped and a median biodiversity impact of land use is calculated. The second method calculates absolute reductions in regional species richness (*A-Regional*) due to land use based on species-area relationships (de Baan *et al.* , 2013b), accounting for the effects of habitat quality of the land use matrix on species loss (Koh & Ghazoul, 2010a). The method calculates the potential regional loss of species within WWF Ecoregions due to the past conversion of habitat and allocates this loss to the different types of land use occurring in each Ecoregion. Land use in regions with little remaining natural habitat or high species diversity receives higher impacts than that in intact or species-poor ecosystems. Both methods assess impacts at a relatively coarse spatial resolution (biomes and Ecoregions, respectively), not necessarily matching the local heterogeneity of biodiversity and the scale of conservation concerns.

Here, we present a new approach to assess the product-related impacts of land use on biodiversity on a grid-cell level. We used species-specific habitat suitability models (HSMs) developed by the Global Mammals Assessment (GMA; Rondinini *et al.* , 2011a) to calculate mammal richness per 900 m grid cell. To assess the impacts of human land use, we modeled the local species richness of a reference scenario and compared it with the richness of human-used land. Species losses were weighted by the level of threat to the species and their global rarity. This new method was applied to a case study of East Africa (5.1), an extremely diverse region hosting multiple Global 200 priority Ecoregions at a vulnerable, critical, or endangered conservation status (Olson & Dinerstein, 2002) and of high richness of mammal species threatened by agriculture (Visconti *et al.* , 2011).

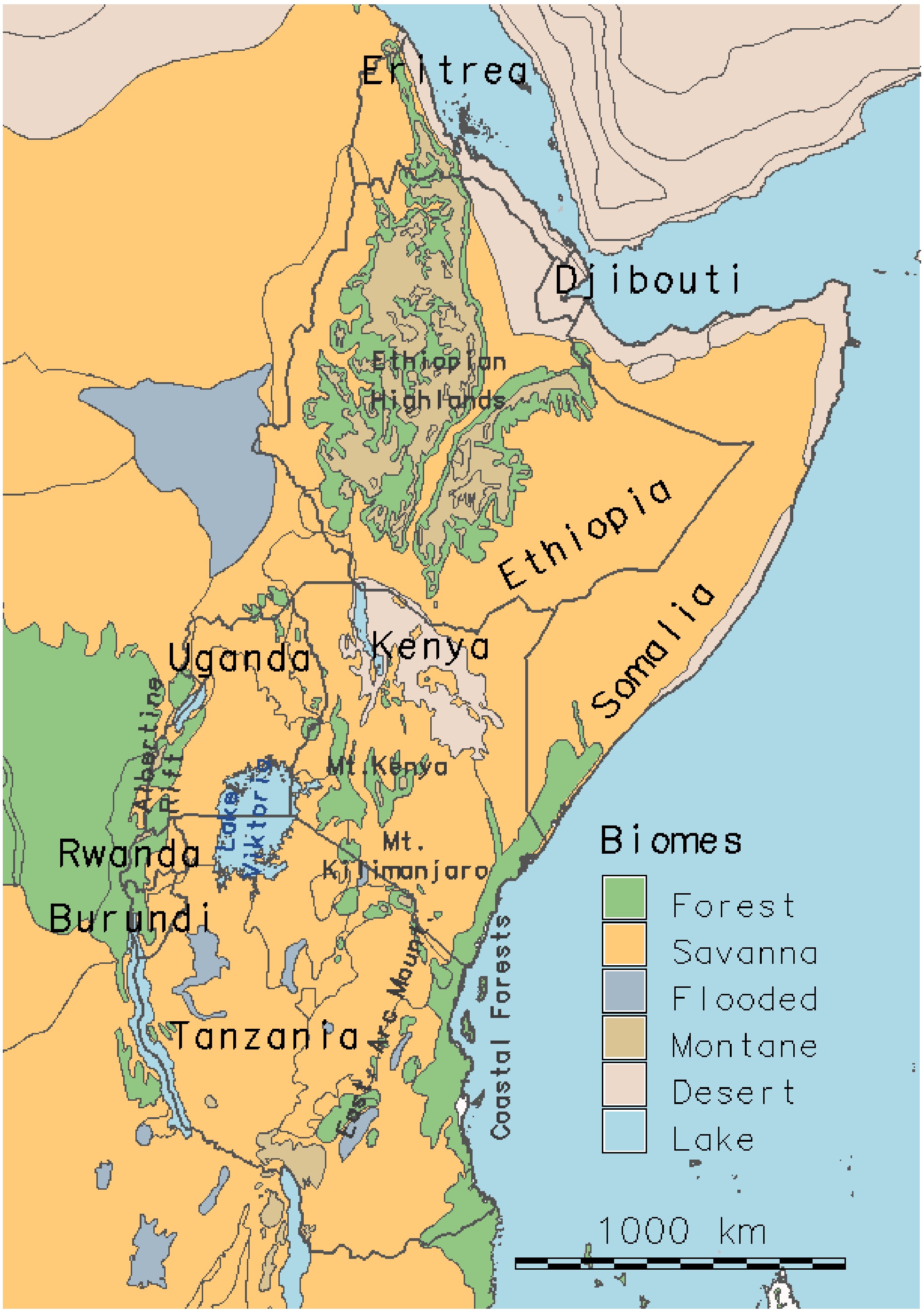

Figure 5.1 – Map of the case study region East Africa, showing country and Ecoregion borders.

Biodiversity impacts of three major export crops (tea, coffee, and tobacco) produced across East Africa were quantified with this method. In addition, biodiversity impacts of these products were assessed with the two previously developed methods described above (de Baan *et al.* , 2013b,a) to compare the outcomes of the modeling approaches.

5.2 Methods

5.2.1 Basic structure of LCA

According to the norms of the International Organization for Standardization (ISO, 2006), LCA consists of four basic steps. First, the goal and scope of an analysis are defined, including the system boundaries of the analyzed product system. To illustrate our land use methods, we considered the biodiversity impacts of land used during crop cultivation (i.e., farm-gate impacts) of three products (tea, coffee, and tobacco). In addition, we considered the impacts of direct land use change driven by the expansion of these crops. The method could, in theory, also be used to quantify indirect impacts (such as deforestation due to wood cutting for tobacco curing or tea drying) or be combined with assessments of other biodiversity impacts (e.g., freshwater eutrophication from waste water of coffee processing or downstream impacts of crop irrigation; Verones *et al.* , 2013a). Impacts were assessed per kg of harvested product. Second, all inputs and emissions of the production systems were compiled in the inventory analysis. Here, we only considered the direct land use and land use change related to crop cultivation (see 5.2.2). Third, in the impact assessment, we assessed the biodiversity impacts of land use per kg of harvested product using three alternative assessment methods. Following an LCA convention, we separately assessed the impacts of land use (occupation) and land use change (transformation) (Koellner *et al.* , 2013b; Mila i Canals *et al.* , 2007). Occupation impacts quantify how much biodiversity is lost as a result of avoiding natural recovery of biodiversity during the actual land use phase. Transformation impacts account for the reduction in biodiversity after (hypothetical) future land abandonment and the time lag until biodiversity is fully recovered (see Appendix 8.4.1). Fourth, the results were interpreted in relation to the goal and scope of the study.

5.2.2 Life cycle inventory analysis

Data for land occupation in m^2.years/kg harvested crop were calculated as the inverse of the yield (based on the data of Monfreda *et al.* , 2008) per grid cell (globally available at a 5 min ~10 km resolution). As climatic conditions above 2500 m asl are not very suitable for commercial production of the three crops, we excluded these high-altitude areas. Land use change (transformation) was calculated per country based on FAO statistics (FAOSTAT, 2013a,b), adapting the approach suggested by Milà i Canals *et al.* (2013)(see Appendix 8.4.4 for further details).

5.2.3 Impact assessment methods

Three potentially globally applicable life cycle impact assessment methods were applied to the case studies (for an overview see 5.1). A description of the local relative method *R-Local* (de Baan *et al.* , 2013a) and regional absolute method *A-Regional* (de Baan *et al.* , 2013b) is given in Appendix 8.4.2 and 8.4.3. The newly developed weighted local loss (*W-Local*) method is described below.

Weighted local loss.

This method was based on HSMs that have been developed by the GMA for nearly all terrestrial mammal species (Rondinini *et al.* , 2011a). Within the species' known geographical ranges, they classified each grid cell as highly suitable (primary) habitat, moderately suitable (secondary) habitat, or unsuitable habitat for that species. In this study, moderately suitable habitat was considered as unsuitable, as the species can be found there but cannot survive there permanently (Rondinini *et al.* , 2011a). The habitat suitability relationship for each species was derived from the 2009 IUCN Red List of Threatened Species based on land cover (based on GlobCover v2.3; Bontemps *et al.* , 2011), elevation, and hydrological features (Rondinini *et al.* , 2011a). The model was implemented in GRASS GIS (GRASS Development Team, 2012).

Calculation of weighted species loss. For the case study region of East Africa, we resampled the original HSMs of all 631 mammal species present from a resolution of 300 m to 900 m to decrease the volume of data. We then estimated potential species richness S per grid cell by summing the presence of each species as predicted by the HSMs. To assess the impacts of human modification of land, we developed two reference scenarios for species richness of all grid cells and compared these with the richness of human-used land. First, the maximum species range reference scenario (*W-Local-Max*) assumed that all areas within the species range were initially covered by a suitable land cover type. Only areas outside the species elevation range or water requirements were considered as unsuitable (e.g., for riparian species, areas that were not in the vicinity of water bodies were excluded from the range). Second, the current land cover (given by GlobCover v2.3; Bontemps *et al.* , 2011) was used as a reference (*W-Local-Cur*). This scenario considers the impacts of changing the land from its current state to future agricultural use. For both reference scenarios, we reran the HSM to derive the potential species richness per grid cell.

We compared these references to two land use scenarios, which represented the potential species richness assuming the cell was occupied by (a) cultivated land with $> 70\%$

Name and reference of method	**Relative local loss (*R-Local*): de Baan *et al.* (2013a)**	**Absolute regional loss (*A-Regional*): (de Baan *et al.* , 2013b)**	**Weighted local loss (*W-local*): developed in this paper**
Biodiversity impact model	meta-analysis of local biodiversity surveys	matrix-calibrated species–area relationships	species-specific habitat suitability models
Scale of impact	local	regional	local, weighted with global threat and rarity
Geographic coverage	global	global	East Africa
Resolution	biome	Ecoregion	pixel (0.81 km2)
Indicators	relative local loss of species richness	absolute regional loss of species richness	weighted absolute local loss of species richness
Taxa	several taxa	mammals (also available for 4 additional taxa)	mammals
Land use classes used in this study (additional classes covered by the study)	annual crops, permanent crops (agroforestry, used forests, secondary forests, artificial areas, pasture)	agricultural land (pasture, used forests, artificial areas)	cropland (> 70%), mosaic cropland (20–70%); (could be extended to any other GlobCover class)
Reference states	(semi)-natural land	natural land	natural land (maximum habitat, W-Local-Max) and current land cover (W-Local-Cur)
Implemented conservation target	maintain high local richness	avoid regional extinction	avoid global extinction

Table 5.1 – Comparison of the four land use methods applied in the case studies

cropland (GlobCover class 10, applied to the annual crop tobacco) or (b) 20–70% mosaic cropland (class 20, applied to the permanent crops tea and coffee).

In the next step, we weighted the species richness S per grid cell by each species' global rarity (R) and threat level (TL), adapting an approach of Verones *et al.* (2013a). Rarity was calculated as the proportion of total global suitable habitat area of the species in the cell (i.e., ranging from 0 to 1). The TL is a rescaling of the categories defined by the IUCN Red List of threatened species (Mace *et al.* , 2008). To give equal weight to the two factors, TL was also scaled from 0 to 1 (0.2—least concern, 0.4—near threatened, 0.6—vulnerable, 0.8—endangered, 1—critically endangered), which deviates from Verones *et al.* (2013a). For each combination of one of the two reference (*ref*) and one of the two land use scenarios (LU_i), we calculated the weighted species loss based on the TL and R of each species k occurring in a cell.

$$S_{W,loss,j} = \sum_{k=1}^{n} TL_{k,ref} * R_{k,ref} - \sum_{k=1}^{n} TL_{k,LUi} * R_{k,LUi} \tag{5.1}$$

Biodiversity impact of land use and land use change. Biodiversity impacts (referred to as characterization factors in LCA terminology) per m^2 of land occupation for each land use type i and reference scenario *ref* were calculated by dividing the weighted species loss per grid cell by the grid cell area (810,000 m^2).

$$BI_{W,occ,LUi,j} = \frac{S_{W,loss,LUi,j}}{810'000m^2} \tag{5.2}$$

The biodiversity impacts of land use change (transformation) were calculated by multiplying the biodiversity impacts of occupation by half the time required for the recovery of biodiversity, t_{reg} , see also Eq. S6:

$$BI_{W,trans,LUi,j} = BI_{W,occ,LUi,j} * 0.5 * t_{reg} \tag{5.3}$$

For t_{reg}, we used estimates for the recovery times of mammals following habitat disturbance from Curran *et al.* (2014). For each grid cell, we modeled recovery times based on the amount of natural habitat in the region, distance to the nearest natural area, latitude, and elevation (see Appendix 8.4.5). The resulting recovery times, which were applied to all mammal species, are displayed in Appendix A.8.2. As a comparison, biodiversity impacts were also calculated based on unweighted species loss (appendices, Figure A.8.6).

5.3 Results

The expansion of tea, coffee, and tobacco cultivation observed in the past 20 years in East Africa are dramatic, with up to a 60% increase in harvested area per country (Appendices, Table A.8.1). No consistent patterns were observed in the spatial distribution of yields or cropland expansion across the three crops (5.2). Extensive land use changes were associated with both tea and coffee in Ethiopia and with tobacco in Tanzania. In Rwanda, the forested area increased in the past 20 years, and permanent cropland decreased. Therefore, no land use change was assigned to any of the three crops. The lowest yields (and therefore the highest land use per kg of crop) were found in Burundi for tea, in Uganda and Ethiopia for coffee, and in South-West Tanzania for tobacco. The land use per kg for tea, coffee, and tobacco was in similar ranges.

When assessed with the rarity and threat level weighted loss (*W-Local-Max*, 5.3), tea, coffee, or tobacco that was produced within the extent of occurrence of very range-restricted mammals had by far the highest impacts on biodiversity. The highest land use impacts occurred northeast of Lake Victoria and in the Albertine Rift Valley (the presumed extent of occurrence of Hopkins's groove-toothed swamp rat, *Pelomys hopkinsi*), in the Ethiopian highlands (Ethiopian striped mouse, *Muriculus imberbis*), in southwestern Ethiopia (Yalden's desmomys, *Desmomys yaldeni*), and southwest of Mount Kenya (ultimate shrew, *Crocicura ultima*). The biodiversity loss assessed with *W-Local-Cur* shows the regions where future expansion of crops would cause the highest impacts. These were regions with high mammal endemism, such as the Albertine Rift Valley in the Ethiopian highlands, the Eastern Arc Mountains, and the Northern Zanzibar-Inhambane coastal forest mosaic.

The relative local species loss (*R-Local*) method (Fig. 5.4, left) was not correlated with the weighted loss (*W-Local*, see Table A.8.5), but it was correlated with the amount of land used per kg of crop. Regions with high land use and land use change showed the highest impacts. The results of the *A-Regional* (Fig. 5.4, right) partly agreed with those of *W-Local-Cur*, such as high impacts in the Eastern Arc Forests. Crops produced in other Ecoregions in Tanzania showed some of the highest impacts, such as the Ittigi-Sumbu thicket, the Serengeti volcanic grassland, and the East African halophytics. The montane moorlands in Kenya, Uganda, and Rwanda also showed high impacts with the *A-Regional* method, as well as coffee produced in the Ecoregion of the East African montane forests.

For all but the *W-Local* method, the transformation impacts of coffee were highest in the southern Ethiopian montane forests. For tea, the Ethiopian highlands also showed high transformation impacts, reflecting the predicted long time lag involved in the recovery in such high-altitude ecosystems (Fig. A.8.2).

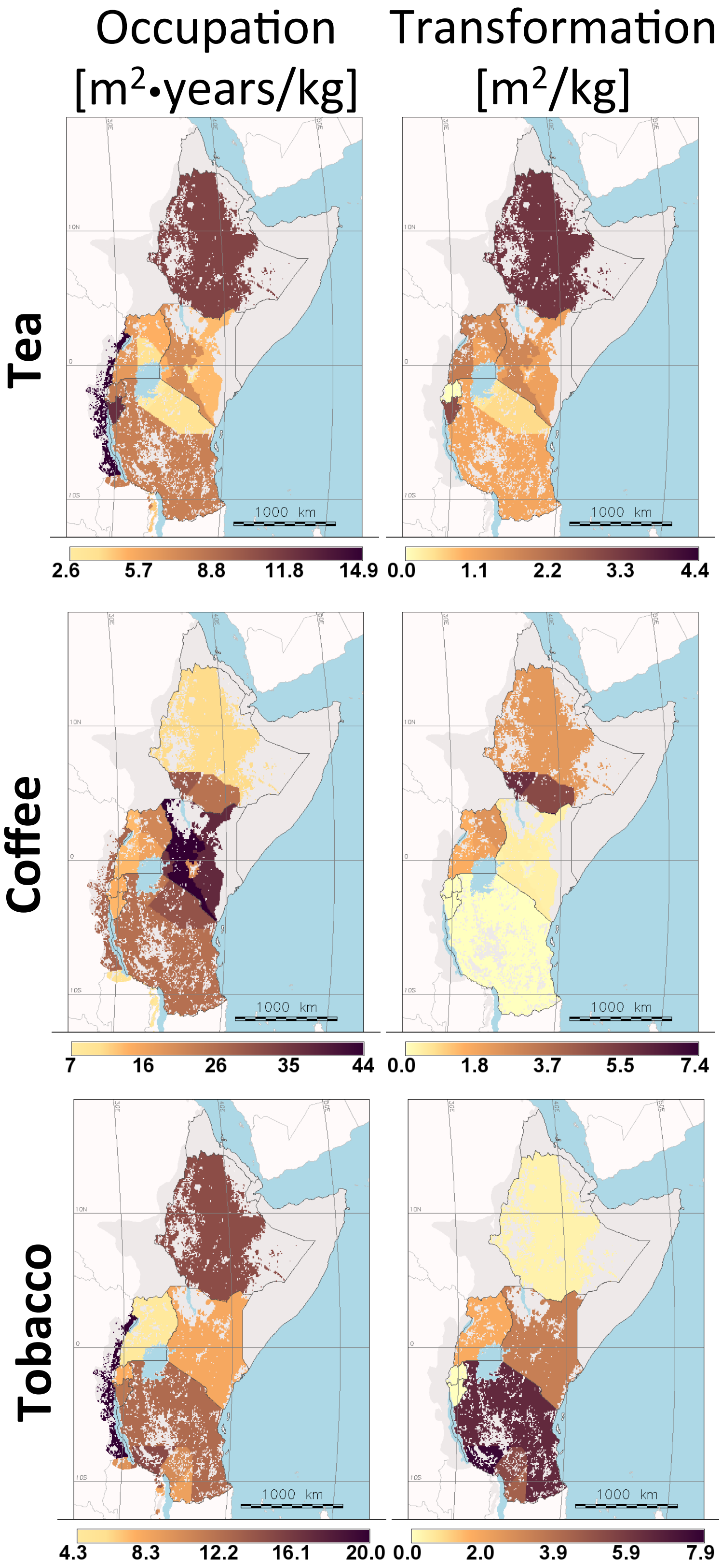

Figure 5.2 – Life cycle inventory. Land use (occupation, m^2*years) and land use change (transformation, m^2) potentially caused by 1 kg of harvested crops.

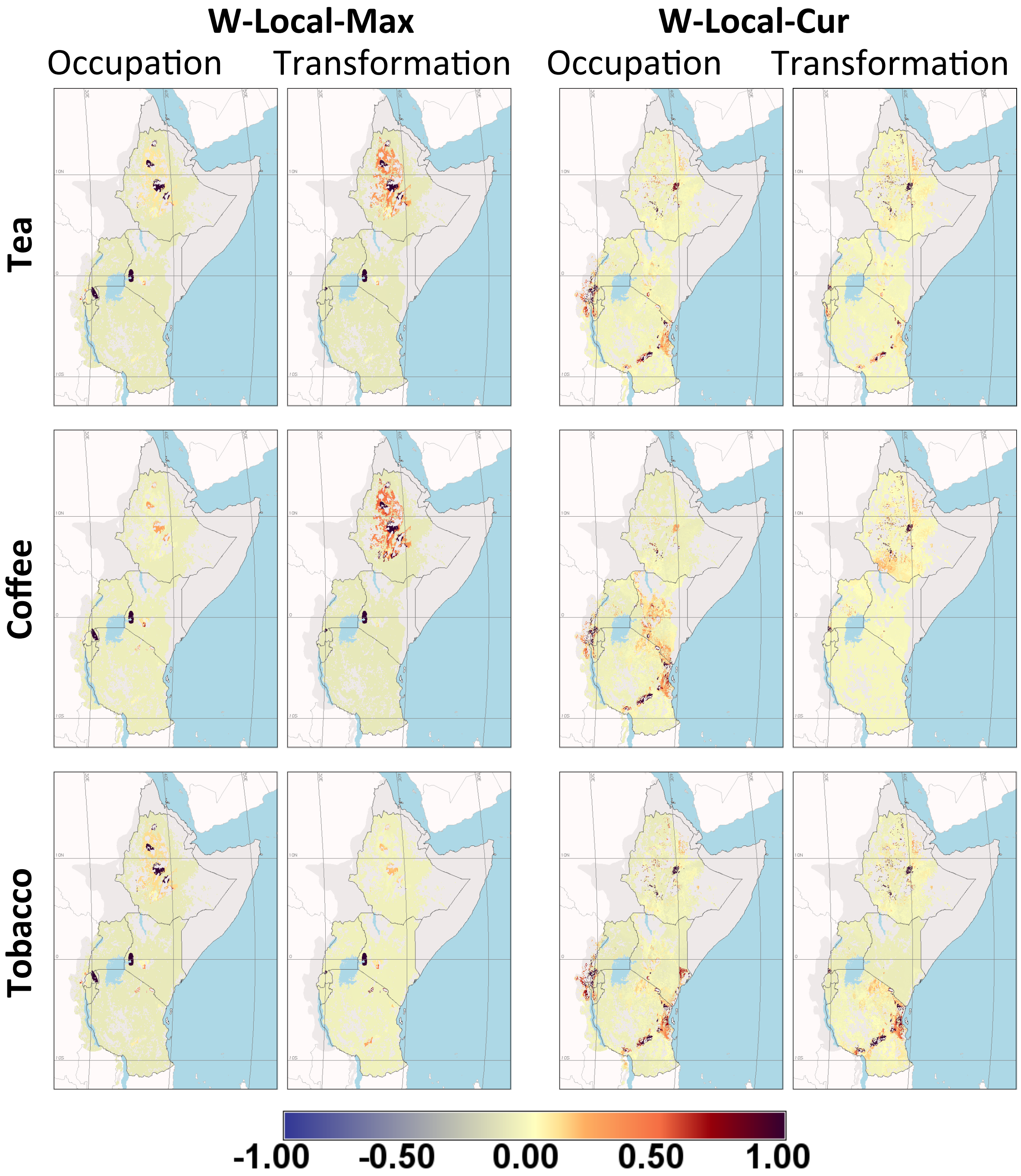

Figure 5.3 – Final LCA result based on the weighted local species loss (W-Local). Biodiversity loss caused by the land use (occupation) and land use change (transformation) per kg of harvested crop. Assessed with the two reference scenarios maximum (Max) and current (Cur). The numbers represent deviations from the mean values of each map (0=mean, -1=one standard deviation smaller than the mean, +1= one standard deviation larger than the mean). Because a few cells had very high values (up to 825 standard deviations above the mean), these values were capped at +1 standard deviation for map display purposes.

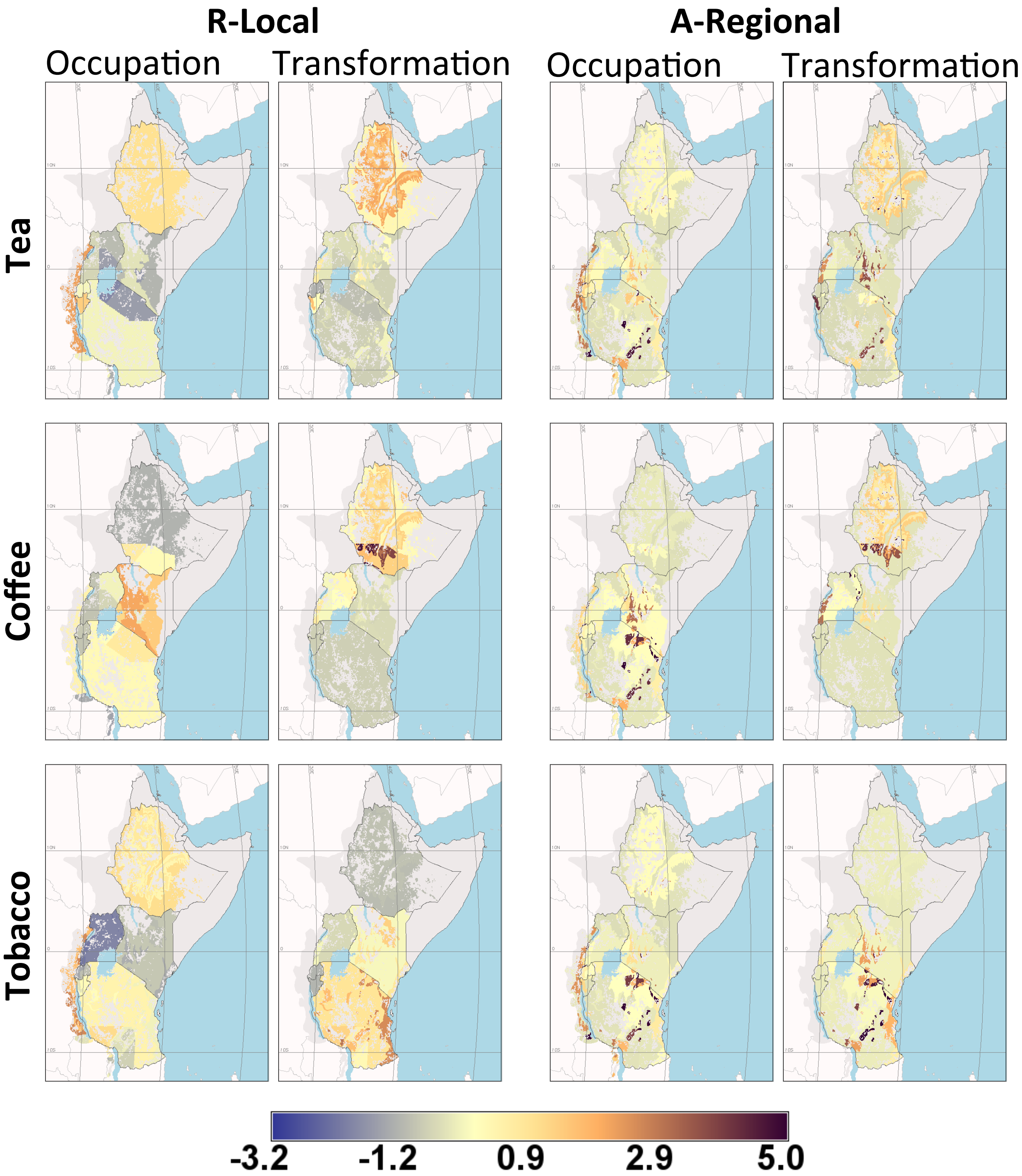

Figure 5.4 – Final LCA result based on relative local (R-Local) and absolute regional (A-Regional) species losses. Biodiversity loss caused by land use (occupation) and land use change (transformation) per kg of harvested crop. The numbers represent deviations from the mean values of each map (0=mean, -1=one standard deviation smaller than the mean, +1= one standard deviation larger than the mean). Because a few cells had very high values (up to 29 standard deviations above the mean), these values were capped at +5 standard deviation for map display purposes.

5.4 Discussion

In this study, we illustrated a novel LCA method to model the production-related biodiversity impacts of tea, coffee, and tobacco across East Africa. The method operates at a high spatial resolution compared to existing LCA land use methods, integrates rarity and threat information, and can be applied to both historical and future impacts of agriculture. We also compared the results to other globally applicable LCA methods and illustrated how different conservation targets can be implemented in LCA studies.

Our results suggest that the potential biodiversity impacts of tea, coffee, and tobacco are most severe in areas of high endemism and low habitat availability for range-restricted species (*W-Local-Cur* maps; Fig. 5.3, right). This agrees with existing conservation assessments of threat, vulnerability, and biological value for the region as a whole (e.g. Burgess *et al.* , 2006). A first step to prevent imminent future extinctions of range-restricted species would be to halt conversion of natural habitat to cropland in high-impact regions (highlighted by *W-Local-Cur*) and to restore habitat in critical areas (highlighted by *W-Local-Max*). The *A-Regional* method highlighted additional areas under high land use pressures and high risk of regional species extinction, where conservation action is most needed.

In the past, East Africa's protected areas were mainly established to protect megafauna in savanna regions, and they are not optimally located to protect small mammals and montane ecosystems (Fjeldså *et al.* , 2004). Protecting these areas is a challenge because they largely overlap with areas of high human population density (Rondinini *et al.* , 2006) and deep poverty (Fisher & Christopher, 2007). The evaluation of socio-economic consequences of land use and conservation planning for poor small-holder farmers (Chiozza *et al.* , 2010) who produce more than half of all coffee and tea in East Africa is thus crucial for successful biodiversity conservation. One potential mechanism to improve the protection of high-impact areas would be to shift the costs of conservation to the consumers of these export-crops in developed countries. A very first step in this direction was observed in the past decade where demand for organic and fair-trade certified coffee and tea strongly increased globally (currently at 16% and 10% of global production, respectively; TCC, 2012; Van Reenen *et al.* , 2010). To achieve effective conservation, a more elaborate framework for biodiversity compensation would have to be developed. The high spatial resolution of the *W-Local* method could serve as a good information basis for the development of such a framework.

To reduce occupation impacts of existing plantations, either the per area impact (e.g., by wildlife-friendly farming practices) or the amount of land used for production (by increasing yields) can be reduced. Which of the two options has the higher potential for biodiversity conservation could not be assessed with the presented LCA methods because they do not provide the necessary detail to distinguish between production systems (e.g., organic and conventional production). If better data on the habitat quality of different production systems, as well as more detailed land cover maps on organic and conventional crop production systems become available, all three methods can, in principle, be expanded (illustrated in the case of milk with the *R-Local* method; Mueller *et al.* , 2014). A more detailed analysis of the relationship between production intensity and biodiversity loss is especially important for coffee where a broad range of production systems exist with very different degrees of impacts on biodiversity (Hundera *et al.* , 2013; Donald, 2004).

The results of the *R-Local* and *A-Regional* were strongly dependent on the land use and the land use change caused per kg of harvested crop. These data were derived by merging FAO crop production statistics with satellite-derived land cover data Monfreda *et al.* , 2008, both of which are subject to considerable uncertainties. The spatial and the temporal variability of yields are only partly reflected in the data. In addition, the available data did not allow us to assess the true spatial patterns of land use change. Instead, we allocated national land conversion rates to all crops produced anywhere in the country. Despite these limitations, we believe that the approach is a promising first step that could be expanded beyond the presented case studies to other common crops and world region.

5.4.1 Conclusions

Traditionally, assessing biodiversity impacts at the product level using LCA relies on linear, static, and nonspatially explicit modeling. In contrast, biodiversity often responds to pressures in nonlinear, dynamic, and site-specific ways. The recently developed methods applied in this paper overcome some of the limitations of traditional LCA models by linking product-specific impacts to different conservation targets. The *W-Local* method can be directly linked to the Aichi Target 12 (CBD, 2013) and the *A-Regional* method indicates the risk of regional (but not yet necessarily global) extinction of species, providing earlier warning signals of extinction. In addition, it can inform on species loss of other taxonomic groups than mammals (i.e. birds, plants, amphibbians and reptiles; de Baan *et al.* , 2013b). The *R-Local* method provides the weakest link to concrete conservation targets. We therefore recommend using the *W-Local* and *A-Regional* method in conjunction. For many LCA studies on internationally traded commodities, the spatial detail on the origin of products required for the *W-Local* method is not available, because of the limited traceability of products along value chains. As an intermediate solution, the *A-Regional* method could be used as stand-alone assessment, using probabilistic approaches to determine the origin of a particular crop.

All three methods can be applied to other regions and crops because they are based on globally available data. We hope this encourages conservation scientists to integrate life cycle thinking in their work and paves the way for interdisciplinary collaboration to better understand the drivers of biodiversity loss and to find solutions for the global biodiversity crisis.

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Chapter 6

Prospects for the international application of biodiversity offsets

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Abstract

Habitat loss in developing countries, primarily driven by agricultural production for developed and emerging markets, represents the most immediate threat to global biodiversity. One expanding policy for dealing with local trade-offs between biodiversity loss and land development is through compensation mechanisms based on the *polluter pays principle*, in particular "biodiversity offsets". Offsets have been proposed at the international level to address displaced trade related impacts, but without any concrete methodological proposals. We develop this idea into a conceptual framework to address land use impacts on biodiversity from agriculture. We illustrate application to a hypothetical scenario of improved conservation in Central Kenya compensating impacts to biodiversity assessed at various spatial scales in East Africa (simulating different grades of data availability on the location of production/impact). First we develop a biodiversity "currency" to quantify conservation value based on threat- and rarity-weighted mammal richness for both a potential natural reference and current land cover state. A land use impact assessment stage quantifies marginal changes in conservation value between potential and current states for different land use systems. To compensate impacts, a spatial conservation planning stage is used to target compensation projects under uncertainty of the location of impact. Following an project planning and evaluation phase (e.g. to measure additionality, efficiency, equity etc.), expected conservation gains of the project are quantified using the same "currency" of conservation value (weighted richness). Offset "ratio multipliers" are used to express risky, uncertain or delayed gains in conservation value in net present terms, to be compared to land use impacts. This allows compensation requirements (in ha of project) to be estimated which can also be expressed as costs per area or agricultural product if data on conservation costs, agricultural yield and product prices are available (e.g. FAO yield and production data). Our results suggest impacts could be compensated with an upper range of price premiums on agricultural products of 10%–28% of consumer prices in the Global North (depending on whether conservation goals are pursued alone or in combination with development aims). These results fall within existing "willingness-to-pay" ranges in the literature. However, our results are highly dependent on context, assumptions of the discount rate applied to future conservation value and other methodological choices within the framework.

6.1 Introduction

In the face of intensifying divers of biodiversity loss and ecosystem degradation, available global funding for the preservation of biodiversity is woefully inadequate (Balmford *et al.* , 2004; McCarthy *et al.* , 2012). Biodiversity is a global public good, the consumption of which is largely non-excludable and non-rival, meaning property rights cannot be easily established and one persons enjoyment does not affect that of others (Farley, 2009). Maintaining adequate provisioning of a public good requires government intervention, either through improved regulation or market-based instruments. However, a "global collective action" problem (Balmford & Whitten, 2003) prevents the necessary resources being mobilized and deployed (analogous to other global public good issues such as climate protection and the regulation of international fisheries). In recent decades, the traditional reliance on regulation (e.g. PAs) has given way to a wide range of novel conservation instruments, particularly the market-based instruments and incentives to preserve biodiversity on private lands (Hanley *et al.* , 2012; Miteva *et al.* , 2012).

Market-based instruments shift the costs of conservation to private actors, scaling payments proportional to either the benefits received or the harm caused (Farley *et al.* , 2010), referred as *beneficiary pays* and *polluter pays* schemes, respectively (TEEB, 2011, Ch. 7). Beneficiary pays schemes attempt to capture the willingness to pay of those who draw the greatest benefits from biodiversity. These include donor-driven multilateral transfers via the Global Environment Facility (GEF), which represents global beneficiaries of positive cross-border externalities, or novel international markets for ecosystem services, such as Reducing Emissions from Deforestation and forest Degradation (REDD). The beneficiary pays principle is appropriate when property rights over a resource (or pollution sink) reside with the polluter (i.e. they are exercising a right to pollute; Kemkes *et al.* , 2010). If not, as with a global public good such as biodiversity, then the polluter pays principle is more appropriate. For biodiversity, this would involve mandating compensation from those who cause biodiversity loss, thereby threatening the provisioning of its benefits to other actors or future generations (e.g. ecological taxation or ecological compensation schemes; TEEB, 2011, ch. 7).

Compensation policies for ecological and biodiversity-related impacts are growing globally. A family of measures termed "biodiversity offsets" are increasingly employed under different auspices in different world regions (e.g. endangered species banking, wetland mitigation and tradable development permits; ten Kate *et al.* , 2004). They focus on compensating local development impacts via habitat trades between an *impact site* and an *offset site*. At the offset site, habitat is either created through ecosystem restoration, or loss is "averted" through improved protection (Maron *et al.* , 2012). The ratio of conservation value gained to value lost is quantified using "offset ratios", based on an integrative habitat quality or service index (e.g. Parkes *et al.* , 2003), which is converted to habitat area ratios (i.e. hectares gained to hectares lost). Offset ratios may, but usually do not, include additional corrections (multipliers) for time delays in the gain of conservation value, future uncertainties, the risk of restoration failure, and the spatial configuration of habitat in the landscape (Bruggeman *et al.* , 2005; Moilanen *et al.* , 2009a). A current lack of ecological rigour in setting offset ratios and multipliers has lead to numerous calls for a substantial overhaul of the methodological framework, e.g., to incorporate time discounting of future biodiversity "gains" (Moilanen *et al.* , 2009a; Curran *et al.* , 2014), info gap analysis to address uncertainties (Moilanen *et al.* , 2009a) and the alignment of offsets with existing regional conservation priorities (Pouzols *et al.* , 2012; Moilanen, 2012a).

Despite the unresolved methodology, offsets and other polluter pays approaches have been strongly promoted in the national and international policy domain (TEEB, 2011, Ch. 7). There have even been calls for a global compensation mechanism to mitigate land use-driven biodiversity loss associated with heavily traded, high-impact commodities (Peterson *et al.* , 2011). Agriculture, as a *proximal* driver of biodiversity loss, is well studied in the literature, generally for single products or production systems (e.g. palm oil, Fitzherbert *et al.* , 2008; coffee, Perfecto *et al.* , 2003; cattle ranching, Alkemade *et al.* , 2013; and plantation forestry, Barlow *et al.* , 2007a). Yet the link between local effects and often distant, *underlying* causes has only recently come to the fore. In particular, the major role of consumption patterns (in developed nations) and increased international trade in driving displaced impacts (Shandra *et al.* , 2009a,b; Lenzen *et al.* , 2012; Gibbs *et al.* , 2010). In a recent systematic assessment of global commodity trade flows, Lenzen *et al.* (2012) estimates that roughly 30% of global species threats are caused by international trade. Perhaps unsurprisingly, many wealthy countries of the Global North exhibited displaced biodiversity impacts to poor Southern countries. This represents a case of *"ecologically unequal exchange*" between North and South (Hornborg, 1998b), which refers to the sale of raw materials and other primary products from relatively poor countries at prices that do not include compensation for local or global externalities (including biodiversity loss).

One response, proposed by Lenzen *et al.* (2012), is to strengthen international regulations for high-impact commodities by expanding the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) to cover commodities linked to species threats. An alternative approach, which we explore here, is to apply the polluter pays principle in the form of a mandatory ecological tax or voluntary premium on traded commodities (e.g. a global value added tax; Farley *et al.* , 2010). In theory, this explicitly links displaced causes (consumption) with local effects (biodiversity loss) via price signals, thereby providing incentives to change production patters (TEEB, 2011, Ch. 7). If payments were used to fund conservation interventions in the region of production, and required compensation was scaled to the estimated magnitude of impact (i.e. applying the "no net loss" offset requirement), this could provide a long-term source of conservation funding. However, such an internationalization of biodiversity offset policy multiplies the challenges already faced by local offsets. These include a lack of a proposed methodology to quantify the impacts of traded goods, that is relatively simple, globally applicable and ecological meaningful. The poor resolution and quality of trade data required to identify the origin of production would also hinder the targeting of compensation activities to match the region of impact (at best, trade data is available at the national scale in multi-region trade databases or publicly available databases of the Food and Agriculture Organization). Finally, the global availability of biodiversity data (species records, checklists, distribution models etc.), and their spatial and temporal resolution, is also generally poor in relation to land cover maps and other data on biodiversity threats (Jetz *et al.* , 2012).

In this paper, we make a first attempt at conceptualizing how how such a scheme could be designed and implicated. In the next sections, we first describe a generic conceptual framework based on an adaptation of concepts from the offset and conservation planning literature. We then illustrate the framework in a hypothetical case study of land use biodiversity impacts and conservation planning in East Africa. Using this case study, we pre-emptively assess expected ecological benefits and economic costs across a number of spatial scenarios with varying parameter estimates and basic assumptions. Finally, we discuss our findings with respect to key methodological issues, benefits/limitations of the approach, along with economic and institutional aspects of a foreseeable implementation.

6.2 Methods

6.2.1 Conceptual framework

Our framework adopts a simplified scenario that consumption in the Global North causes a displaced biodiversity impact (via trade) in the Global South, which is compensated through a conservation activity at or near to the impact location (Figure 6.1). The framework represents a simplified international adaptation of recent research on offsets that combines techniques in offsets with those of spatial conservation planning, uncertainty and risk assessment (Overton *et al.* , 2013; Moilanen *et al.* , 2009a; Moilanen, 2012a; Pouzols *et al.* , 2012). The framework is divided into five major clusters of analysis steps, which we describe below briefly, before describing the specific details of our case study and the data we used:

1. **Biodiversity currency**. Offsets require defining a biodiversity “currency” (ten Kate *et al.* , 2004). The currency is any measure of marginal conservation value (Overton *et al.* , 2013) suitable to quantify loss and gain between impact and offset site. In spatial conservation planning, conservation value is quantified by transforming data on raw biodiversity “features” at a site (e.g. species, vegetation communities) using a “biodiversity benefit function”. The benefit function defines how features at a site are weighed and aggregated (e.g. by inverse range size or conservation threat status) and may incorporate feature response to habitat change (e.g. land use) and conservation interventions, or current conservation status

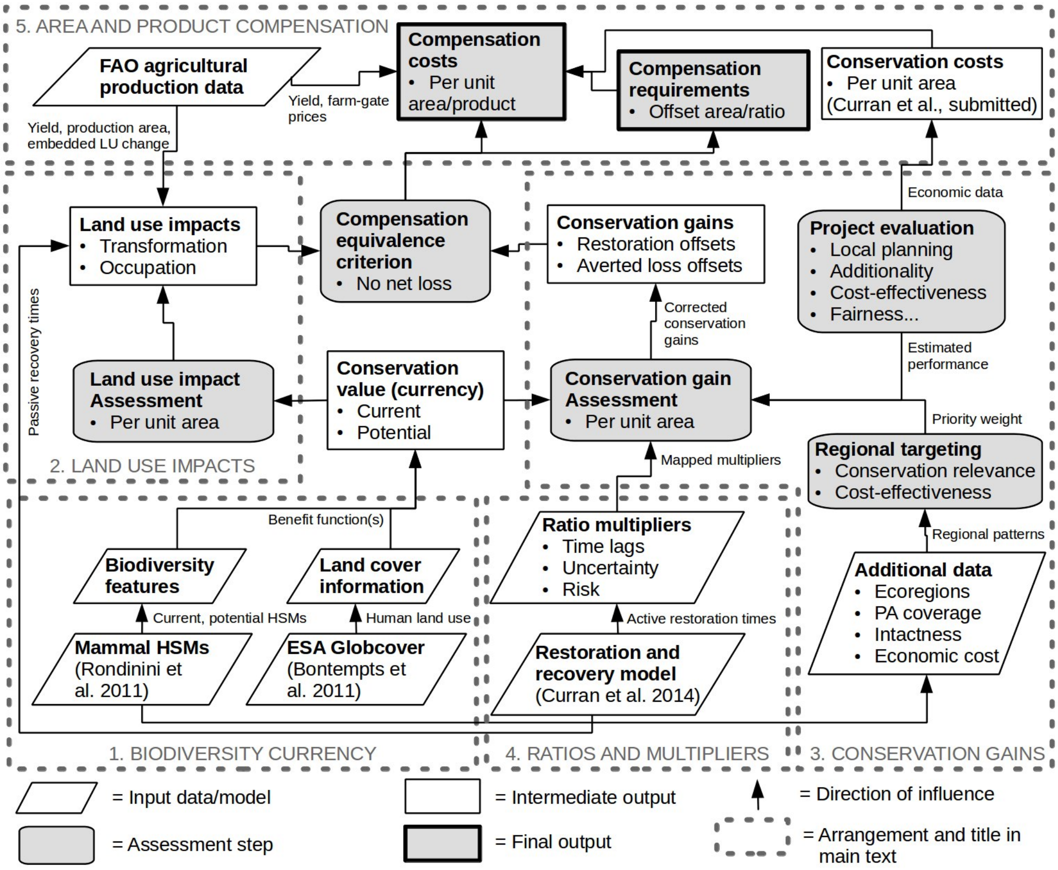

Figure 6.1 – **Conceptual compensation framework.** Assessment steps group together into five clusters (dotted grey boxes and titles surrounding elements in the figure): **1) Biodiversity currency**. A measure of marginal conservation value derived from applying biodiversity benefit function(s) to biodiversity feature data, calculated for potential (reference) and current states; **2) Land use impacts.** Land use impact assessment quantifies impacts of land use as change in biodiversity currency between potential and current states; **3) Conservation gains.** To compensate impacts from land use, an "equivalent" amount of conservation value, expressed in the same currency, must be created in the approximate region of production (i.e. impact site). Where impact location is unknown, the offset is weighed by a regional conservation relevance weight from a prioritization assessment. A project evaluation step is also assumed to establish baselines, assess equity/efficiency/additionality etc.; **4) Ratios and multipliers.** Expected conservation gains involve time delays (e.g. habitat restoration), risks and uncertainties, accounted for using "ratio multipliers" resulting from time-discounting and uncertainty analysis; **5) Area and product compensation.** An equivalence criteria (e.g. no net loss) is used to calculate compensation requirement (area protected or restored) and costs (i.e. $/ha or % price increase of products). References for published or submitted datasets and models used in the case study is also indicated in the relevant location in the diagram.

relative to protection targets (Pouzols *et al.* , 2012; Moilanen *et al.* , 2009b). In our case study (see below), we used biodiversity feature data (habitat suitability models) from a recent global mammal assessment (Rondinini *et al.* , 2011a) for potential (reference) and current land cover scenarios (current scenarios use a recent land cover dataset).

2. **Land use impacts.** Biodiversity loss may arise through a number of impact pathways (e.g. land use and land use change, pollution via petrochemical fertilizer and pesticide use, water stress due to irrigation requirements), but our framework and case study focus on land use and land use change impacts of agricultural and urban land. However, the framework could be adapted to other drivers provided information on intensity, exposure and biodiversity response is available.

3. **Conservation gains.** Compensation of biodiversity impacts via biodiversity offsets requires that an "equivalent amount" of biodiversity value is created in the approximate region of production (i.e. the "impact site" under offset terminology). Biodiversity value is said to be "created" when degraded/converted habitat is restored (*restoration offsets*) or biodiversity loss is averted (*averted loss offsets*) through improved protection (Maron *et al.* , 2012). If the impact location is unknown or spatially uncertain (as in our case study), we assume the compensation activity is targeted using a spatial conservation planning approach (e.g. Moilanen *et al.* , 2009b). A project evaluation step is then required to establish baselines and success criteria, set strategies and targets, monitor performance and assess equity/efficiency/additionality implications (e.g. Miteva *et al.* , 2012; van Oosterzee *et al.* , 2012; Wuenscher & Engel, 2012).

4. **Ratios and multipliers.** Expected conservation gains generally involve time delays (e.g. habitat restoration), risks and uncertainties. In the case of habitat restoration, time lags in the maturation of conservation value in secondary habitats can be substantial, ranging tens to hundreds (or even thousands) of years and involve a substantial risk of outright failure (Maron *et al.* , 2012; Curran *et al.* , 2014; Moilanen *et al.* , 2009a). While habitat protection may be more certain, it also involves risks regarding the future maintenance of biodiversity in a reserve (protection may be inadequate to prevent encroachment, habitat loss elsewhere in the landscape could undermine the viability of local populations, political regimes or zoning laws may change etc.). Time discounting is often used to account for such temporal uncertainties and risks, and may be applied to both impact and gain in diversity, possible using a range of positive or negative rates (see Figure 6.3). Discounting, uncertainty and risk assessment are used to develop offset "ratio multipliers" that adjust expected conservation gains downwards depending on the

time and risk profile of the conservation project. Ratio multipliers convert the total expected biodiversity gains of a project to discounted *present* gains for both restoration or averted loss offsets (in our case study, we use an empirical model of biodiversity restoration and recovery; Curran *et al.* , 2014).

5. **Area and product compensation.** Adjust conservation gains express expected conservation value in net present terms (termed Net Present Biodiversity Value by Overton *et al.* , 2013). These can be compared to estimated present (or past) losses per unit area (i.e. median impacts per ha of agricultural compared against median gains per ha of the compensation project). An equivalence criteria (e.g. no net loss) is used to assess compensation requirements (area protected or restored) for each unit of impact and the likely costs based on the data from the compensation project. Our rationale is that compensating one year of impacts of agricultural production impacts requires funding one year of conservation of similar amount conservation value in the context of a long term compensation project. If additional data on land use productivity and product prices are available, the price increases associated with compensation can be estimated for different agricultural products or land use systems.

6.2.2 Case study overview

We illustrate our framework with a hypothetical case study with analyses at different scales in East Africa. For the case study, we assumed a landscape-level conservation project located in Central Kenya is being assessed in terms of the potential to compensate impacts from agricultural land use in East Africa (Fig. 6.2). We adopted three levels of uncertainty in the land use impact assessment with regards to knowledge on the location of production (the impact site): *Regional*, within East Africa; *National*, within Kenya; and *Sub-national*, within Central Kenya (an area spanning 17 districts around the Aberdare Conservation Area and Mount Kenya). At the regional scale, we delineated the extent of analysis along ecological rather than political borders, following the boundaries of WWF “Ecoregions” (Olson *et al.* , 2001) occurring within case study countries (Burundi, Djibouti, Eritrea, Ethiopia, Kenya, Rwanda, Somalia, Tanzania and Uganda), judging this to be more relevant from a conservation perspective. The national scale assumes product origin is known at the country level, which is perhaps most likely for most globally traded commodities (although see Monfreda *et al.* , 2008 for sub-national, spatial predictions of FAO data). Finally, the sub-national scale assumes the origin of the product is known or can be estimated at a higher degree of accuracy (e.g. important areas for agricultural export, such as Central Kenya).

We constructed conservation scenarios at the smallest sub-national scale simulating a landscape-level conservation project aimed at compensating impacts at the three scales listed above. In the next sections, we describe the data and analyses used for the steps in our conceptual framework (i.e. establishing a biodiversity benefit function, conducting a land use impact assessment, targeting and evaluating the project, developing multipliers and assessing the cost of compensation per unit of developed land/product). In our case study, the data and analyses used to represent these steps are often highly simplified. Given the wide scope of the research, trade-off off's with accuracy and precision are unavoidable. Our main aim is to present *one* possible interpretation of an "internationalized" offset framework to prospectively identify key issues and influential variables related to ecological effectiveness and economic costs. We are not proposing a fixed methodology, recommending any form of implementation, or assessing the social fairness and ethical considerations of market-based instruments and the commodification of nature (see Pascual *et al.* , 2010; Kosoy & Corbera, 2010).

6.2.3 Biodiversity "currency"

In local restoration and averted loss offsets, habitat *area* is most frequently used as the "currency" for habitat trades (e.g. hectares lost to hectares gained ten Kate *et al.* , 2004). Area ratios are generally weighted to account for local habitat quality/condition, diversity or species-specific factors (Parkes *et al.* , 2003; BBOP, 2012). Recent work has also integrated spatial considerations in the placement of compensation areas to account for landscape level processes and patterns (e.g. Bruggeman *et al.* , 2005). In general, applying tools from spatial conservation planning can be used to include such considerations using an explicit biodiversity benefit function to define the currency, and analyse the equity of trades (Moilanen, 2012a). We adopted this latter approach to develop a simple biodiversity currency based on threat- and rarity-weighted mammals species richness.

Biodiversity feature data. We used habitat suitability models (HSMs) from a recent Global Mammal Assessment (GMA) for 575 extant mammal species occurring in the study region with available habitat suitability information (Rondinini *et al.* , 2011a). The mammal data represent a modification of expert-based IUCN range maps (i.e. each species' known geographic extent of occurrence) to highlight suitable pixels of habitat based on land cover associations, elevational range limits and water-dependence for each species, using data from the 2009 IUCN Red List of Threatened Species (Rondinini *et al.* , 2011a). We refer to these data as *current* HSMs, highlighting approximately current patterns in suitable habitat for each species.

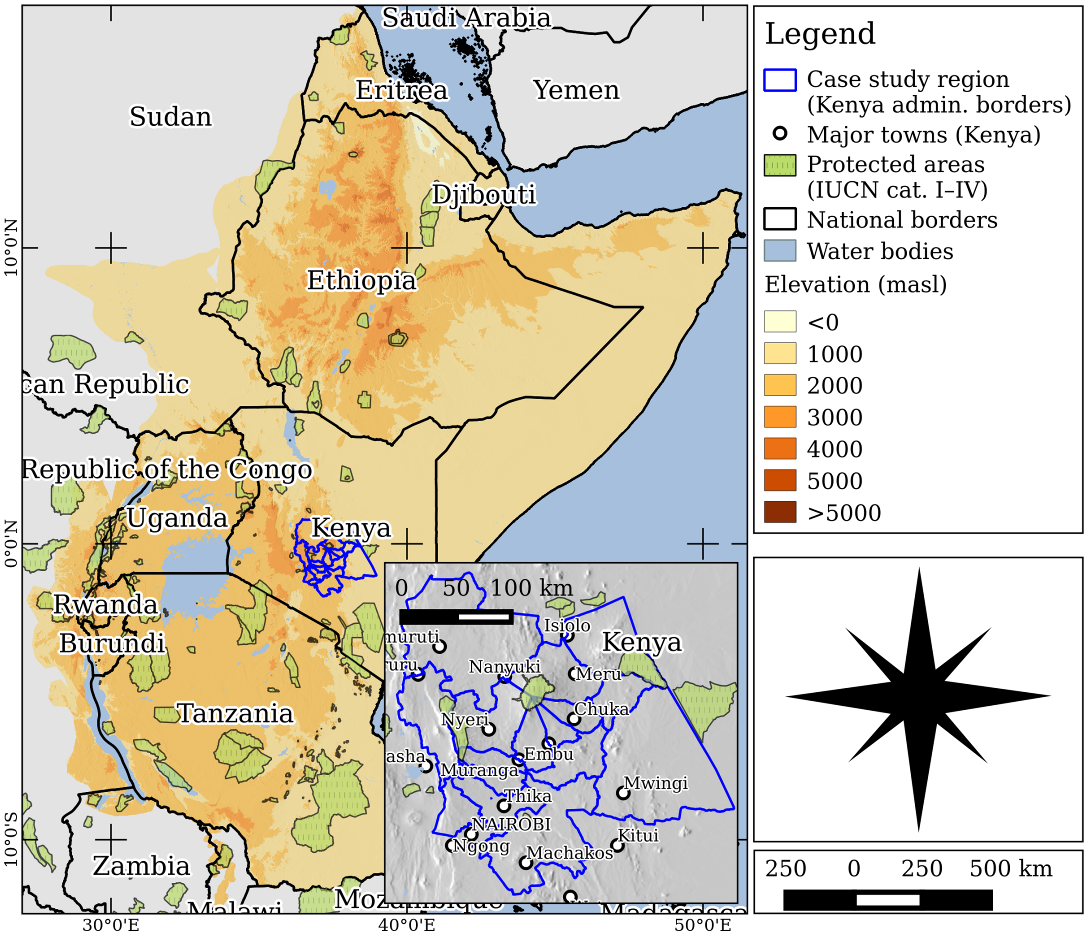

Figure 6.2 – Map of the case study region showing elevation and protected area coverage. The largest scale of analysis (regional) followed Ecoregion borders within East Africa. The national scale analysis was applied to Kenya (national); and the smallest scale covered a hypothetical conservation landscape in Central Kenya (inset map with blue administrative borders). Elevation and hillshade maps derived from SRTM 90m digital elevation model. PA coverage from the World Database on Protected Areas 2010 release (UNEP-WCMC, 2010).

To quantify the impacts of land use on biodiversity, a choice of baseline or reference condition is required (Koellner & Scholz, 2007). Our reference was based on the "maximum species range reference scenario" of de Baan *et al.* (n.d.), which assumes that all pixels within a species' known range were initially covered by a suitable land cover type, subject to elevation and hydrological constraints (i.e. re-running the GMA filtering procedure using only elevation and water-dependence criteria). We refer to these data as *potential* HSMs, emphasizing the desire to characterize patterns in biodiversity in the absence of human land use.

Benefit function. We used a simple biodiversity benefit function to translate changes in species composition brought about by marginal land use changes or conservation interventions to to marginal changes in an index of conservation value. The benefit function was developed by de Baan *et al.* (n.d.) as a variant of *weighted richness* (*WR*) of mammals, expressed as:

$$WR_i = \sum_j w_j q_{ij} \tag{6.1}$$

Where $w_j$represents the weight of species j and q_{ij}represents the proportion of the global distribution of species j in cell i (the proportional *range equivalent* of the species in the cell). Weights can be any attribute that differentiates the conservation value of a species, including threat level, economic or cultural value etc. Conservation value is thus derived from both the richness of a cell, the relative rarity of the component species in the cell and their respective weights for other factors. As an additional weight to range size, we used each species' IUCN threat status (Mace *et al.* , 2008), rescaled from 0 to 1 (0.2: least concern, 0.4: near threatened, 0.6: vulnerable, 0.8: endangered, 1: critically endangered; IUCN vers. 3.1, second edition). Applying this benefit function, our biodiversity "currency" (used to quantify loss and gain) was thus expressed in units of weighted species range equivalents (*sp. eq.*).

Current and reference biodiversity states. We calculated weighted richness for both the potential (reference) and current HSMs. For the main analysis, the weighing factors for both potential and current benefit function reflected the species' current range and IUCN status (i.e. the weighing of each species under the "potential" baseline reflects *current* patterns of rarity and threat). When assessing a loss or gain of biodiversity, this assumes only *marginal* changes in conservation value within a static modelling context (i.e. that any land use impact or conservation gain does not significantly change global patterns in species' rarity or threat). Therefore, the method could not be used to assess loss attributed to very large-scale or historic changes in land cover, because these would significantly change species' range equivalents used in the benefit function. Thus, for such a situation, a dynamic scenario modelling approach would be required that adjusts rarity and threat information (conservation value per pixel) as land is converted/set aside for conservation (e.g. Pouzols *et al.* , 2012). We denote this approach WR_{marg} in the text.

To investigate how the results change under a different set of assumptions, we also repeated the analysis using a different benefit function with weights reflective of *historic* changes in weighted richness. In this case, the weight for range size was calculated separately for both potential and current biodiversity models (i.e. the potential model used rarity weights from the *potential* HSMs, and the current model from the current HSMs), and no threat weight was applied. This approach is denoted WR_{hist} in the text, and assumes that in the reference (pre-disturbance), biodiversity is plentiful and threats are absent. As habitat becomes scarce, the marginal conservation value of remaining habitat patches increases non-linearly, reflecting the increasing rarity of the component species. Species that have seen a large reduction in their range due to anthropogenic land use thus receive a much higher weight in the current state than in the potential baseline.

A source of uncertainty for both weighing schemes is the potential HSMs which only consider elevation and water range restrictions to species occupancy, not potential vegetation patterns (de Baan *et al.* , n.d.). Thus habitat suitability and occupancy rates are overestimated compared to the current HSMs, which use species habitat preferences based on the ESA Globcover 2009 land cover product (vers 2.3; Bontemps *et al.* , 2011). While the use of potential vegetation maps for the region could improve this step (e.g. P. van Breugel, pers. comm.; Kindt *et al.* , 2007b), this was beyond the scope of the research (and would restrict applicability to other world regions where such maps do not exist).

6.2.4 Land use impacts

Impact assessment framework. To assess losses of conservation value, we adopted a land use assessment framework drawn from the field of Life Cycle Assessment (LCA). We only briefly describe the framework here, but see Koellner & Scholz (2007); Koellner *et al.* (2013b); de Baan *et al.* (n.d.) for a comprehensive description. In LCA, two types of land use impacts are recognized: *transformation* impacts and *occupation* impacts, which are illustrated in the context of the general offset model in Figure 6.3. Transformation impact reflect the initial land use change (conversion) of natural habitat to an anthropogenically-modified state (e.g. the logging of a forest). Without a proceeding land occupation period (e.g. agriculture), local biodiversity would slowly recovery to a state comparable to the pre-transformation state, all else being equal. The hypothetical time lag in the recovery of biodiversity is attributed to the initial transformation in a space-for-time substitution (i.e. land transformation impact = biodiversity reduction·area·recovery time). Therefore converting 1 ha of habitat that takes 100 y to recover is equivalent to converting 100 ha of habitat that takes 1 y to recover (Koellner *et al.* , 2013b). The use of "recovery times" in this context represents both a weighing factor to penalize land use change in ecosystems predicted to recover very slowly, and an accounting step to correctly allocate causality of long-lasting, future impacts to the initial activity that caused them.

Assuming land is occupied for a human use following habitat conversion (e.g. to produce crops), the recovery of biodiversity is continuously suppressed. This suppression of conservation value represents the land use *occupation* impact, and is calculated as biodiversity reduction·area·occupation time. For our hypothetical case study, we mapped pixel-wide patterns in WR_{marg} across the region for both potential and current HSMs (at

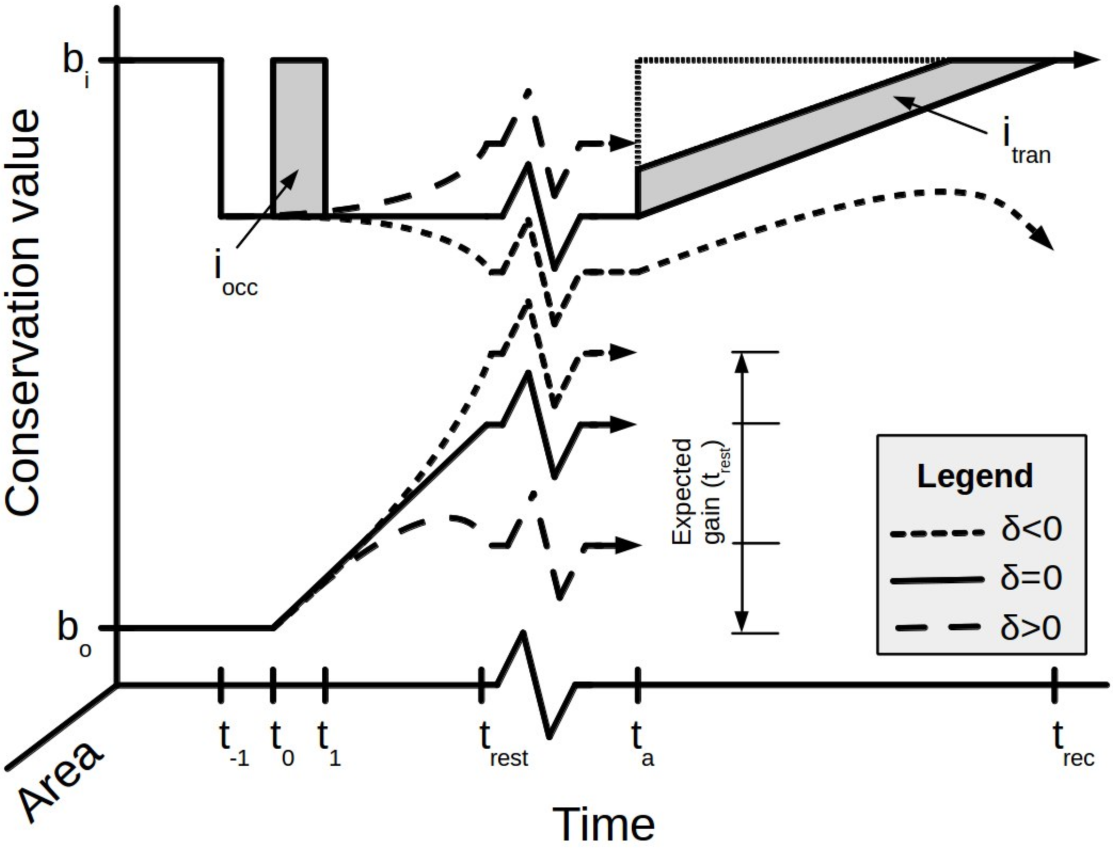

Figure 6.3 – General restoration offset model showing various options to discount conservation value (positive, δ>0; neutral, δ=0; and negative, δ<0) for the impact and offset sites. Changes in biodiversity value are shown for the impact site (b_i) and offset site (b_o) over time. For our land use impact assessment, we illustrate occupation impacts ($i_{occ)}$ and transformation impacts (i_{tran}) as grey boxes in the impact trajectory. Occupation impacts assume conversion of habitat occurs some time in the past (at t_{-1}on the *x-axis*). The offset starts at the present (t_0), where the impact is assessed within a bounded occupation time (t_0 to t_1) representing one year of agricultural production. Transformation impacts would generally make up the entire triangle bounded by the recovery trajectory, after a hypothetical abandonment of used land at some unspecified time in the future (t_a) leading to a passive recovery time (ending at time t_{rec}). In our analysis of transformation impacts (de Baan *et al.* , n.d.), only the proportion of the crop production associated with habitat transformations in the past 20 years is considered (i.e. the grey portion of the transformation triangle making up i_{tran} only partially covers the total transformation impacts). The offset ratio is calculated based on the difference in biodiversity value between impact and offset, which might be estimated at the end of the restoration period (t_{rest}, as in our analysis) or as an average ratio across the time horizon (from t_0 to t_{rest}; also tested in our sensitivity analysis). The third component of the assessment, area, is only symbolically illustrated on the *z-axis* for simplicity.

a resolution of 900m). To quantify land use impacts on conservation value, we calculated the loss of weighted richness between the potential and current models as:

$$WR_{loss} = WR_{potential} - WR_{current} \tag{6.2}$$

Land cover characterization and product-level assessment. We assessed the loss of conservation value (WR_{marg} and WR_{hist}) between potential and current states for anthropogenic land cover classes of the ESA Globcover 2009 land cover product (vers 2.3; Bontemps *et al.* , 2011). This led to marginal biodiversity impacts for 13 land use classes, consisting of seven uniform agricultural classes (LC#10–16), six mosaic classes (LC#20–32) and 1 urban/built land class (LC#190). We calculated summary statistics per land use class at the three scales of analysis detailed in section 6.2.2 (Fig. 6.2; regional, national and sub-national), eventually using the median impact, lower and upper quartiles (25 and 75 percentiles) per cell for each land use class in our calculations (due to the non-normal skew of the impact data).

To allocate impacts at the product level (i.e. per tonne of commodity), and to estimate changes in farm-gate prices caused by compensation payments, we used agricultural production, yield, price and trade data from the FAO for the year 2010 (FAOSTAT, http://faostat.fao.org/). For the local Kenya case study, we used Kenya-specific data. For the regional analysis we used average data across countries. To estimate average yield and price information for aggregate land cover classes (e.g. LC#10: "Managed and cultivated areas"), we used a production weighted average of yield and price to represent a likely mix of crops for that region. This led to estimates of yield (tonnes/ha), prices ($/tonne) and value added ($/ha) per "commodity equivalent" for each land cover class and country in the case study region (see Appendix 8.5.1 for details).

6.2.5 Conservation gains

Regional targeting of compensation activities

In our case study, and in contrast to existing biodiversity offset applications, we assumed that knowledge on the spatial location of the impact site (i.e. production site of an agricultural commodity) is highly uncertain. We assumed basic knowledge was only available at three spatial scales (regional, national and sub-national), thus a spatial targeting system is required to allocate compensation activities among competing options. This is essentially the same overarching goal as in spatial conservation planning, where different alternatives for investing conservation resources are ranked based on biodiversity feature, cost and other management data within a static modelling context (Pouzols *et al.*

, 2012). To illustrate this step, we used the conservation prioritization software *Zonation,* vers 3.1 (Moilanen *et al.* , 2009c; Moilanen, 2007) to develop a regional conservation prioritization map of East Africa. *Zonation* iteratively partitions (or *zones*) a landscape to maximize representation of biodiversity features while minimizing costs.

Biodiversity feature data and analysis settings. To carry out the conservation prioritization, we used the same mammal habitat suitability models (Rondinini *et al.* , 2011a) as input data, rescaled to 1'800 m (due to computational limitations). We included ecosystem representation based on WWF Ecoregions (Olson *et al.* , 2001), a habitat condition layer based on simplified ESA Globcover land use classes, and a negatively-weighted layer of distance to nearest semi-natural habitat or protected area to promote reserve agglomeration (semi-natural habitat was defined as all non-anthropogenic Globcover classes). *Zonation* provides a range of biodiversity benefit functions to quantify changes in marginal conservation value associated with reserve expansion. We used the additive benefit function in a target-based planning approach, aiming for 25% coverage of each species' regional range, forcing inclusiong of all existing protected in the final reserve network (see Appendix 8.5.4 for details on the input data, *Zonation*'s additive benefit function and the target based planning approach).

Regional conservation costs. To represent economic costs of conservation, we used data on the total economic value of agricultural production from the Global Agro-ecological Zones (GAEZ) database v3.0[1], taking the highest value for each pixel from either (i) all agricultural crops or (ii) cereal crops (Appendix, A.8.1). We used these data to represent conservation opportunity costs (i.e. the value of the next best land use), similar to McCarthy *et al.* (2012). We added to these opportunity costs, estimates of conservation management costs for the "land purchases or easements" strategy of Curran *et al.* (n.d.a) from Kenya (assuming these were applicable to the entire East Africa region). Appendix 8.5.4 contains further details regarding cost categories and estimates.

Project planning and evaluation

To illustrate the project planning and evaluation step, we used a hypothetical landscape-level conservation project in Central Kenya (Figure 6.2). We delineated the landscape extent to include 17 administrative districts encompassing the Aberdare Range and Mount Kenya. The area is ecologically very important for sustaining national populations of large mammals, birds and endemic flora (WRI, 2007), and supports a large share of national agricultural output (e.g. 70% of the country's tea is produced around the

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[1] http://webarchive.iiasa.ac.at/Research/LUC/GAEZv3.0/

Aberdare Range; Rhino Ark, 2011). We chose a very large landscape, such that its size (ca. 90'000 km^2) was sufficient to support viable populations of large vertebrates and resilient expanses of ecosystems on a proportion of the area, consistent with the concept of "conservation landscapes" (Hanski, 2011). While we do not define such a minimum scale criterion for compensation projects, nor include population viability analysis in our conservation planning step, such a criterion would be necessary to ensure that conservation gains are permanent, and not undermined by habitat loss or degradation outside of the project's area of influence.

Local conservation planning. We repeated the *Zonation* prioritization step on a local scale, altering the methods as follows. We used a higher resolution of 900 m, and replaced WWF Ecoregion representation with a 25% representation target of 8 simple vegetation classes created by combining bands of percentage tree cover with elevation bands (see Curran *et al.* , n.d.a for details). We also included an explicit agglomeration function based on the Boundary Length Penalty in *Zonation.* This relative negative weight ranges 0–1, and discourages reserve selection with a high area to perimeter ratio (Moilanen & Wintle, 2007). We used a low BLP of 0.02 after experimentation with different settings. We forced inclusion of existing protected areas in the final reserve network, as well as a eight established wildlife conservancies owned and managed by non-profit organizations and community trusts (data from the Laikipia Wildlife Forum; www.laikipia.org).

Economic cost data at the local scale was based on the land price model and management cost estimates of Curran *et al.* (n.d.a). In a second cost scenario, we added a premium to all operational costs representing additional funding for a development programme focusing on infrastructure improvements, healthcare, water, sanitation, education, and capacity building (development cost data from Curran *et al.* , n.d.a). This scenario investigated how compensation costs would increase when a prior development aim is included in the compensation activities. Finally, to simulate an additional form of financial support for existing protected areas, we developed a third scenario that *additionally* assumed half the predicted operating costs for existing protected areas are included in the compensation project budget. Given the severe shortfalls in protected area financing (McCarthy *et al.* , 2012; Balmford *et al.* , 2003a), additional support for existing PAs constitutes an unavoidable cost for effective future conservation.

Quantifying conservation "gains". To evaluate conservation gains in our simplified case study, we assumed that the solution landscape resulting from the *Zonation* analysis (i.e. reserve map to encompass at least 25% of the distributions of all species and vegetation classes) was implemented through expansion of existing reserves. Gains in conservation value were assessed for both averted loss and restoration offsets under the prioritization map. We used the same biodiversity "currency" of weighted richness as in land use impact assessment, applying both WR_{marg} and WR_{hist} adaptations to the benefit function.

For averted loss offsets (i.e. improved protection of existing natural habitat), gains were assumed to represent only the portion of conservation value threatened by loss. Thus we omitted the residual conservation value that persists in farmlands, which was represented by the average conservation value of two anthropogenic land use classes: LC#10 "Cultivated and managed areas" and LC#20 "Mosaic cropland (50-70%) / vegetation (grassland/shrubland/forest) (20-50%)". The data layers required to calculate residual conservation values for for unconverted natural areas were generated by re-running the mammal HSMs assuming that all pixels in the region were converted to LC#10 and LC#20, respectively (see de Baan *et al.* , n.d., for details). For restoration offsets, we assumed the aim was to restore back to the potential reference state, and thus considered the difference between current and potential states for each pixel as the expected "gain" in conservation value (i.e. the inverse of the land use impact).

Project evaluation. In addition to ecological and economic factors, a thorough project assessment would also consider criteria of additionality (counter-factual validation of effectiveness), leakage (effect of displaced impacts outside the project area) and equity (social fairness and costs associated to the project), as well as assessing the legal and institutional conditions necessary for success. However, we heavily simplified this step for our case study because it was both beyond the scope, and not the aim, of our research. There is also an abundance of studies focusing on these issues within the applied conservation (e.g. van Oosterzee *et al.* , 2012; Wunder *et al.* , 2008; Pattanayak *et al.* , 2010; Miteva *et al.* , 2012; Ferraro, 2009) and prioritization literature (e.g. Eklund *et al.* , 2011; Wuenscher & Engel, 2012).

In terms of additionality, we assumed all habitat outside of officially protected areas represented additional conservation gains, and did not consider the intensity of threat or vulnerability of the habitat. We then applied time-discounting to these conservation gains to account for future uncertainty and time delays, differentiating between averted loss offsets, which were discounted over an assumed project time horizon of 100 y, and restoration offsets, which were discounted over the expected recovery time of the habitat in question (see below for details). With regards to equity, our baseline cost scenario maximized cost effectiveness through a strategy of land purchases and easements (data from Curran *et al.* , n.d.a), assuming a "compensation" fairness criterion (i.e. the costs of protection are compensated without any additional equity effect Pascual *et al.* , 2010). However, our development cost scenario illustrated the effect of a different fairness criterion, assuming an *additional* yearly egalitarian payment for development purposes for all landowners and communities affected by the project. For a broader discussion of the equity, policy and institutional implications of different conservation strategies in the region, see Curran *et al.* (n.d.a).

6.2.6 Offset ratios and multipliers

Present value of future conservation gains. To calculate the discounted present value of future gains from habitat restoration offsets, we accounted for both time lags and restoration failure risk. For time lags, we spatially predicted recovery times for species similarity for all pixels in the region using the "general" model in the meta-analysis of Curran *et al.* (2014). This expressed the time required for Sorenson species similarity between recovering and old growth habitats to reach background old growth values under a regime of active restoration. To be conservative, we used the maximum recovery time estimate across taxa (birds, mammals, herpetofauna, insects, other invertebrates, plants, and trees) for each pixel to discount expected conservation value at maturity using an exponential discounting model (see below for choice of discount rate). Discounting at maturity assumes that conservation gains are delivered only after full habitat recovery, and ignores the intermediate biodiversity value of secondary habitats as they recover. To test how results change when relaxing this assumption, we also calculated multipliers that account for this growing biodiversity value by subtracting the rate of recovery (in % per year) from the discount rate, assuming linear recovery of conservation value over time (i.e. annual percentage increase = 100/recovery time).

We separately accounted for a 12% risk of restoration failure, estimated by Curran *et al.* (2014). While this might appear low, failure is defined as a neutral or negative age–similarity relationship, with any positive effect considered an eventual success and dealt with using time-discounting (i.e. a predicted recovery time of 1000 y would be considered a "success" and dealt with using time-discounting, despite the discounted present gains being negligible). A full description of the recovery time predictions, the resulting map, derivation of restoration risk, and the discounting formula is provided in Appendix 8.5.2. For the data underpinning and statistical models, see Curran *et al.* (2014).

Discount rates. We discounted the delivery (restoration) and persistence (averted loss) of expected future conservation gains. We used a rate of 1% for averted loss offsets, and 4% for restoration offsets, based on recommendations of Overton *et al.* (2013). For averted loss offsets, we adopted the simple assumption of Overton *et al.* (2013), that if the biodiversity value between offset and impact sites is equivalent (i.e. in-kind offsets;

represented by habitat area in the that study) and habitat loss within protected areas is zero, then the discount rate equals the rate of habitat loss outside of protected areas (see Appendix, Section 8.5.3). We thus used a rate based on the approximate annual rate of loss of natural habitat in Kenya (averaged across forest and non-forest habitats using data from the FAO; FAOSTAT, 2013a). For restoration offsets, we used a rate of 4%, derived from taking the rate for averted loss (1%) and adding 1.5% for both the "opportunity cost" and "marginal productivity" effects cited by Overton *et al.* (2013). The former represents the pure time delay or direct biodiversity debit caused by the temporal separation of loss and gain. The latter represents the fact that this lost biodiversity would have had a productivity effect (i.e. it would reproduce and create more biodiversity) which has also been temporarily suppressed until the restored areas shifts from being a net sink for immigration to a net source of emigration (we elaborate on these discounting choices in the discussion). Finally, we tested for sensitivity of results to the discount rate by running a subset of the analysis at rates of 1% for averted loss and 2% for restoration, 2% and 4%, and 4% and 8%, respectively.

6.2.7 Area and product compensation

The final step in the conceptual framework consists of assessing the conditions for equivalence in loss and gain at the area and product level, and estimating resulting compensation costs based on data from the compensation project. In our case study, we took the median discounted conservation gain across all cells covered by the compensation project. This represents the present value of expected conservation gains of a cell at the end of the planning/restoration period, essentially a risk-adjusted guarantee that the conservation value of the cell will be retained and/or improved over the specified time horizon. These annual gains can be directly compared to median land occupation impacts (expressed in annual impacts per unit area; see section 6.2.3). The effective offset ratio, R_u, expressed as area offset per area impacted for land use u is:

$$R_u = \frac{\overline{i_u}}{\overline{p_m}} \tag{6.3}$$

Where $\overline{p}_m$ is the median annual conservation gain per unit area for compensation project m, and $\overline{i_u}$ is the median annual impact per area of land use u. The ratio thus expressed the amount of area units required to compensate a single area unit of impact, for a particular compensation project. Because we worked with median data (due to the skew of the data), we did not simply divide total conservation gains by per area impacts, as pixels with extreme values in conservation gain would dominate the calculation. We thus compared median impacts with median gains to be consistent.

Because we could estimate average conservation costs per unit area ($/ha), we calculated the likely compensation costs for a range of land use classes, and modelled predicted farm-gate price changes of agricultural products based on yield and production data from the FAO (see Section 6.2.3 and Appendix 8.5.1 for details). Values at the product level indicate average annual premiums per unit of production-weighted crop mix (crop equivalents) from a specific land use class.

Single-crop assessments: tea, coffee and tobacco. To illustrate a further application of the framework, we also simulated compensating the biodiversity impacts of three specific export crops from East Africa (tea, coffee and tobacco) by applying the "maximum species range" impact assessment method of de Baan *et al.* (n.d.). This method is identical to the approach highlighted in section 6.2.3, but uses additional spatial information on the production and yield of specific crops (from Monfreda *et al.* , 2008) to better link impact assessment results to specific products (i.e. only considering impacts that fall within the predicted distribution of a crop, weighed by its yield). An additional difference is the inclusion of "transformation" (land use change) impacts, as oppose to pure occupation (land use) impacts assessed in this study. This involved accounting for the estimated land use change (conversion of forest habitat) caused by expansion of the specific crop over the past 20 years using FAO data (FAOSTAT, 2013a). This gives an estimate of the production share that originates from recently deforested land (de Baan *et al.* , n.d.; Milà i Canals *et al.* , 2013). To obtain an average annual estimate of the "embedded" land use change of any single crop, we divided this transformation impact by the time horizon (20 y).

Assessing the impact of land transformation (land use change) assumes a space-for-time substitution, where occupation impacts of WR_{marg} (in sp. eq./m^2/y) are multiplied by a hypothetical recovery time (y), defined as the time necessary for biodiversity to recover back to the potential reference state (these estimates were based on the recovery models presented in Appendix 8.5.2, but assuming passive rather than active recovery; see de Baan *et al.* , n.d. for details). Transformation impacts in areas that are slow to recover and high in biodiversity value can therefore be orders of magnitude larger than occupation impacts. While the method makes very simplifying and often subjective assumptions (i.e. only accounts for land use change in the past 20 y, impacts are marginal and fully reversible, time and space are perfect substitutes) it facilitates integrated assessment of both land use and land use (Koellner *et al.* , 2013b; Mila i Canals *et al.* , 2007). For further details on the method, data and impact assessment results, see de Baan *et al.* (n.d.).

6.3 Results

6.3.1 Biodiversity currency and land use impact assessment

Applying the marginal variant of the weighted richness benefit function (WR_{marg}) resulted in a biodiversity currency that was strongly influenced by threatened and rare species. Spatial patterns in conservation value were thus strongly linked to the ranges of such species (potential biodiversity model shown in Fig. 6.4, left). Estimated loss of conservation value (in %) between potential and current models was most pronounced in heavily cultivated areas around Lake Victoria and the Ethiopian Rift complex (Figure 6.4, right).

Land use impacts. Land use-specific impacts on conservation value for the various land use classes of the ESA Globcover classification system were highly variable and influenced by the spatial scale of assessment (Fig. 6.5). At the regional (East Africa) scale, the mosaic forest–agriculture land use class (LC #32) received highest impacts, with a median value of 1.19E-06 species range equivalents (sp. eq.), almost one order of magnitude larger than the average impact across classes (4.21E-07 sp. eq.). At the national and sub-national scale, fewer land use classes were present and impact patterns were less clear (Figure 6.5). Non-forest mixed classes and open irrigated classes (ie. LC#12 “Post-flooding or irrigated shrub or tree crops”, and LC#13 “Post-flooding or irrigated herbaceous crops”) received the lowest impacts.

Alternative species weights. Applying the historic variant of the biodiversity benefit function (WR_{hist}) resulted in a currency that was less spatially variable (Appendix 8.5.5, Fig. A.8.2), due to the alternative weighing system for conservation value between potential and current biodiversity models. Relative patterns of impacts across LU classes and scale were similar, except that LC#31 (“Mosaic grassland or shrubland (50–70%) / cropland (20–50%)”) exhibited the highest impacts, in contrast to lowest impacts using WR_{marg} (see Appendix 8.5.5 for WR_{hist} results).

6.3.2 Regional targeting through conservation prioritization

We used a spatial conservation prioritization approach to simulate the targeting of compensation projects under uncertainty, using the program *Zonation.* Results suggested effective representation of all mammals species could be achieved with ca. 30% of

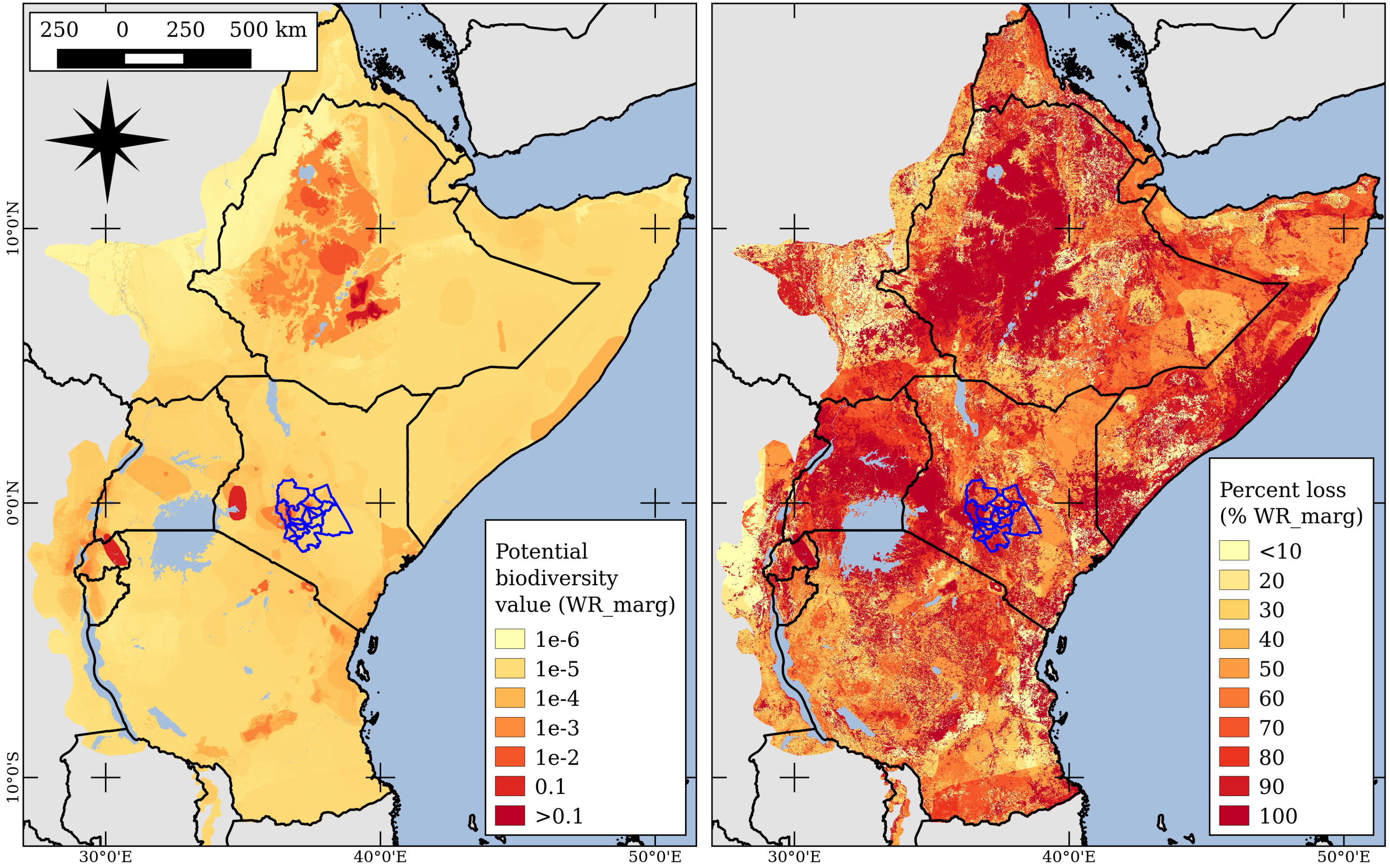

Figure 6.4 – Potential pattern (left) and percentage reduction (right) in weighted species richness using the WR_{marg} benefit function (threat and rarity weighted richness). The potential pattern represents reference WR_{marg} values in the absence of anthropogenic land use. It was calculated using mammal habitat suitability models from the "maximum species range reference scenario" of de Baan *et al.* (n.d.). Percentage loss shows difference between potential and current states, where current states refers to WR_{marg} values calculated based on contemporary habitat suitability models using data from Rondinini *et al.* (2011a).

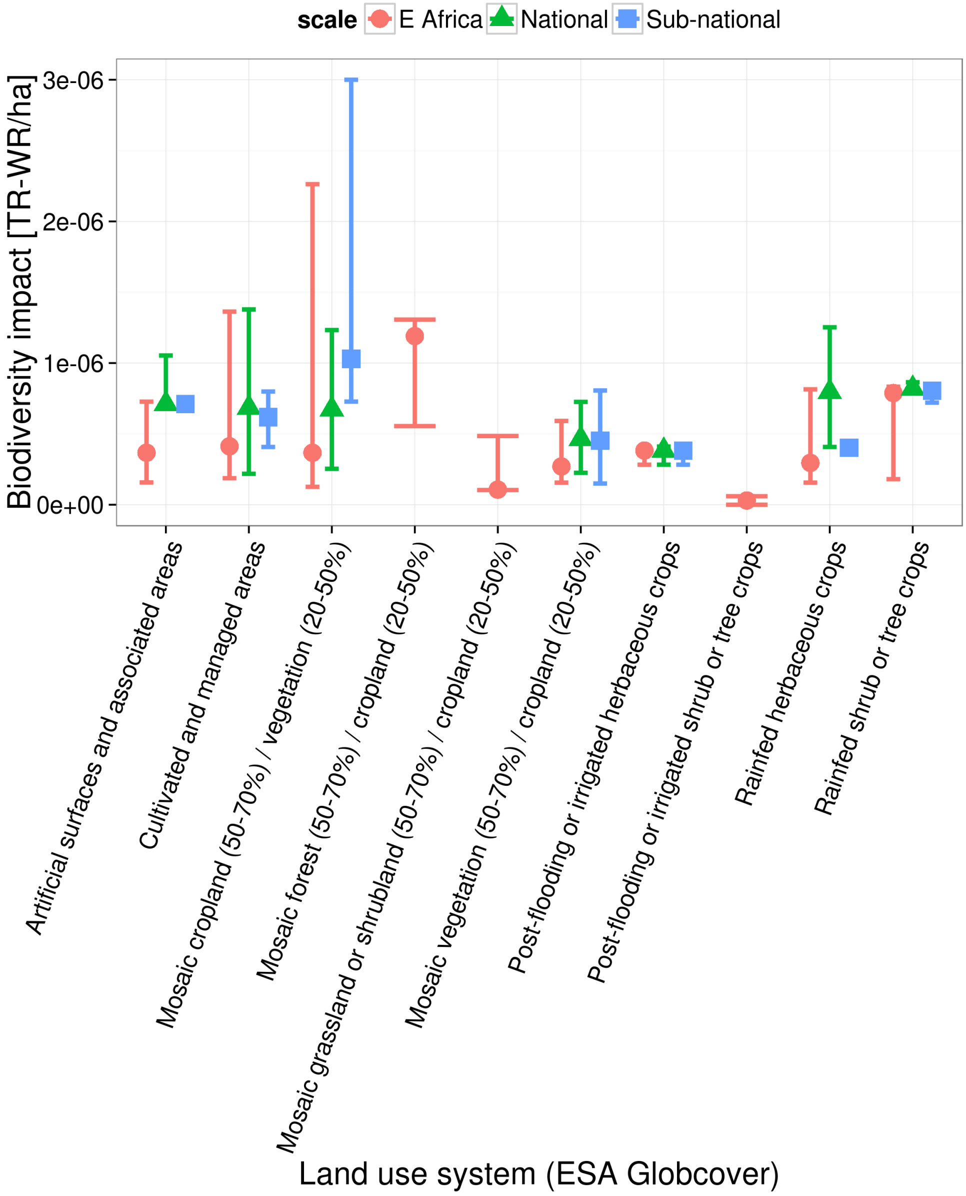

Figure 6.5 – Biodiversity impacts of ten ESA Globcover land use classes at three spatial scales using the WR_{marg} benefit function. Plot shows median impacts and inner quartiles (25 and 75 percentiles) per ha for each land use class. Missing data for specific scales (e.g. "Mosaic grassland..." at national and sub-national scale) indicate no presence of land use class at that scale. TR-WR = threat and rarity weighted richness

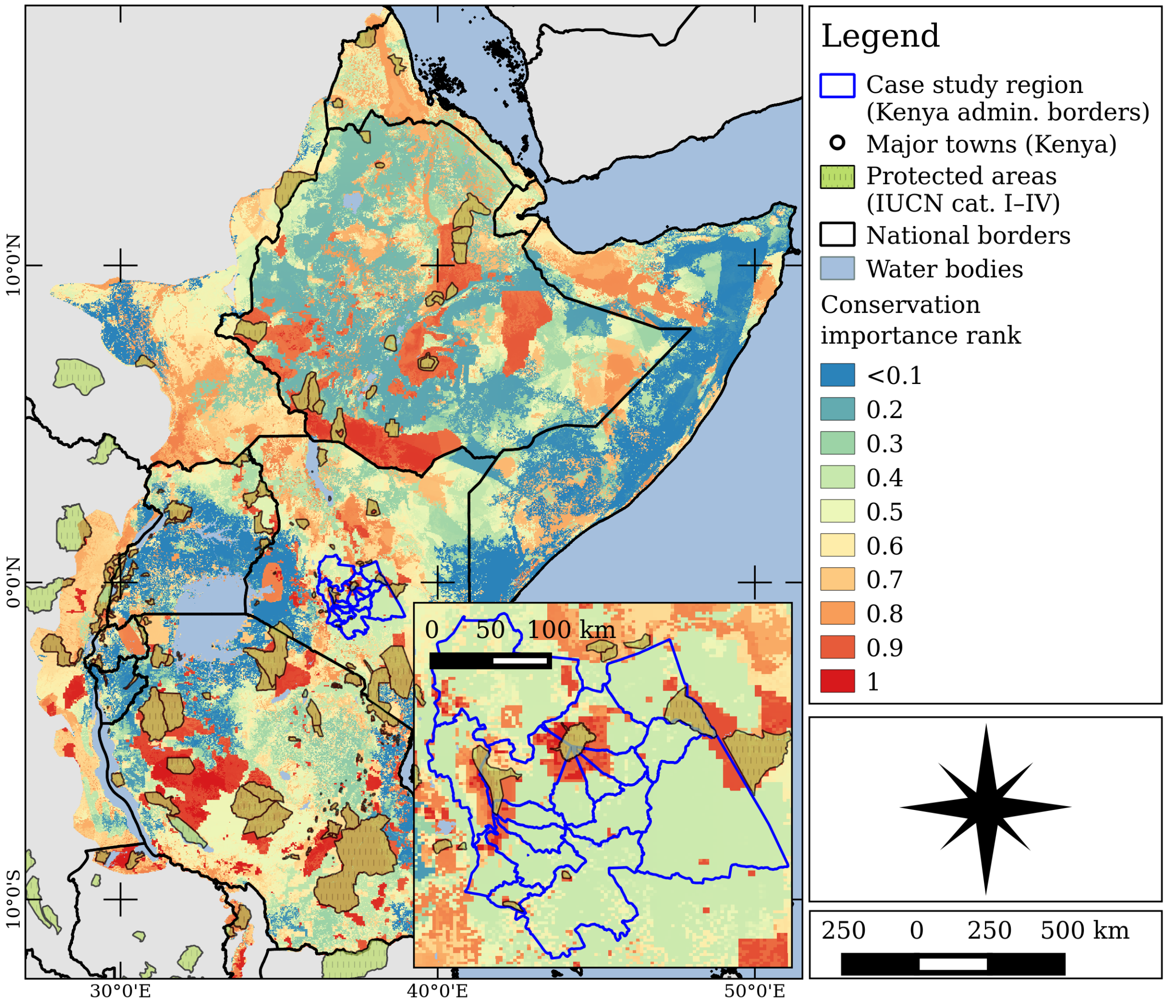

Figure 6.6 – Regional conservation ranking based on *Zonation* target-based planning. Analysis aims to represent 25% of the regional potential ranges of all mammal species and Ecoregions.

the East African landscape set aside for conservation. This would involve expanding existing reserve networks along the border between the Democratic Republic of Congo and Uganda, in South-East Sudan, and in Central/Northern Kenya (Fig. 6.6).

Regional conservation relevance of the compensation project. Only part of our hypothetical case-study landscape in Kenya was in the top ranked (30%) zone for species protection. In a more comprehensive application of our framework, a correction would be required to account for the reduced regional conservation relevance of our case study. One option could be to use a landscape ranking output directly, such as that produced by *Zonation* (Fig. 6.6), as a weighing factor for the conservation gains of compensation projects. Alternatively, projects could be restricted to a threshold value (e.g. the highest 30% of the landscape). However, to illustrate our framework and reduce complexity, we did not make any such adjustments.

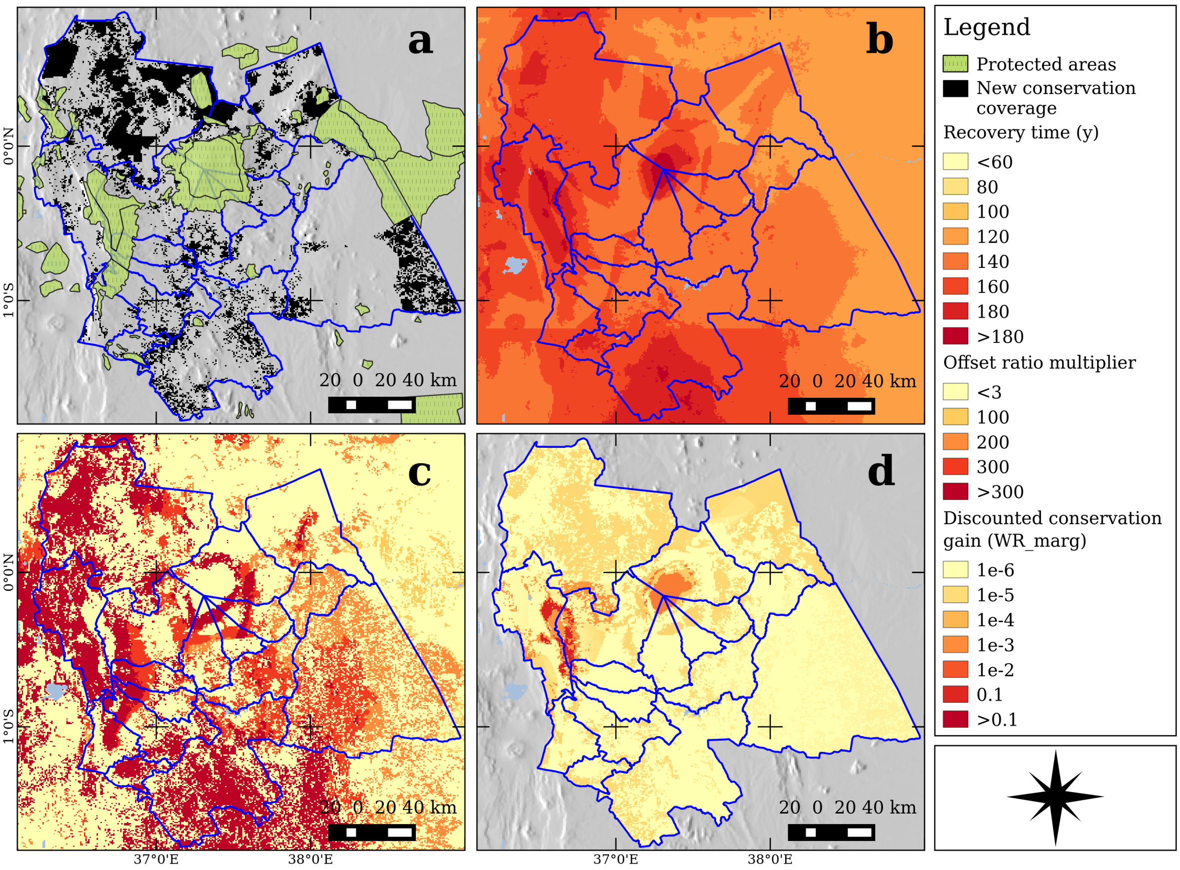

Figure 6.7 – Local *Zonation* target-based planning result showing proposed areas for new conservation coverage (a), recovery time predictions (b), offset ratio multipliers (c) and discounted present conservation gains (c) for the hypothetical compensation project in Central Kenya. Legend for all figures shown on the right of diagram in order of presentation. *Zonation* output achieves 25% regional potential range coverage of all mammal species and vegetation communities. Recover time predictions based on models of Curran *et al.* (2014) for the slowest taxon to recover. Ratio multipliers calculated at discount rate of 1% for averted loss offsets and 4% for restoration offsets across project time horizon (100 y) and recovery times, respectively.Discounted present conservation gains calculated as expected future conservation value divided by the ratio multiplier.

6.3.3 Local planning and project conservation gains

***Local* Zonation.** The local *Zonation* analysis resulted in 34% of the landscape set aside for conservation (including existing PAs) to meet the target of 25% coverage of the potential regional range of each species and vegetation community (Fig. 6.7a). The majority of this coverage was placed on remnant natural habitat fragments (particularly the extensive rangelands and conservancies in the North of the study region), and was mainly composed of existing natural habitat, but with a substantial portion of converted land for habitat restoration and high associated multipliers (Fig. 6.7a and 6.7c). We assessed total conservation costs for the new reserve using a spatial dataset for the region based on land prices and conservation management costs (Appendix A.8.1, Fig. A.8.1). Total estimated net present costs over a 100 y horizon ranged from nearly \$10 to \$34 billion, depending on the cost scenario (Table 6.1).

Conservation scenario	**Coverage [ha]**	**Tot. gain, WR_{marg} [sp. eq.] @ 1%p, 4%r**	**Tot. NPC [M$] @ 1%**	**Unit NPC [K$/ha] @ 1%**
25% sp. coverage, CONS	1’028’295	1.08	9’766.8	9.47
25% sp. coverage, PPA	1’551’150	18.18	16’575.5	10.65
25% sp. coverage, DEV	1’028’295	1.08	22’536.4	21.86
25% sp. coverage, ALL	1’551’150	18.18	33’852.0	21.75

Table 6.1 – Biodiversity gains and net present costs (NPC) for the hypothetical compensation project in Central Kenya. Total biodiversity gains reflect summed pixel values under new conservation area. CONS = only non-PA land considered; PPA = support included for half the management costs of public PA cats 5 & 6; DEV = includes development budget; ALL = includes both development budget and public PA support; M = million; K = thousand.

Discounted present conservation gains. Offset ratio multipliers based on predicted recovery times were very large due to the long predicted recovery times. Recovery times within the compensation project landscape ranged 105–197 y (mean of 142) in already-converted, non-natural pixels (Fig. 6.7a). This translated into ratio multipliers ranging 81–3’445 (mean of 454) for habitat restoration (Fig. 6.7b). For averted loss, the low discount rate of 1% over the set 100 y project time horizon resulted in only modest multipliers of 2.7 uniformly applied to all natural pixels.

For the main analysis, we applied discount rates of 1% and 4% for averted loss and restoration offsets, respectively. Total biodiversity gains from the reserve network constituting the hypothetical compensation project were 1.08 sp. eq. for the marginal loss method (WR_{marg}; Table 6.1) and 2.74E-07 sp. eq. for the historic loss method (WR_{hist}).

Sensitivity to discounting assumptions. To test the sensitivity of results to discounting assumptions, we calculated ratio multipliers that accounted for the intermediate growth of conservation value in secondary habitats over time, rather than only the value at full habitat maturity (see Methods, Section 6.2.6). This resulted in more modest multipliers ranging 25–1088 (mean of 143).

We also tested how multipliers are influenced by the discount rate, applying rates of 2%, 4% and 8% to restoration offsets and 1%, 2% and 4% to averted loss offsets. Doubling the discount rate had a large influence on ratio multipliers, particularly for restoration offsets in areas with long predicted recovery times (Table 6.2). For example, decreasing our rate of 4% to 2% changed the lower range of multipliers from 81 to 9.3, and the upper range from 3’445 to 60 in our hypothetical project landscape. However, restoration offsets concerned only a minority of the area selected for the project, thus moving from 1%/4% for protection/restoration to 1%/2% changed median gains only from 4.6E-06 to 4.3E-06, or a 6.7% decrease (CONS scenario).

Variable	**Discount rates**			
	1%p, 2%r	**1%p, 4%r**	**2%p, 4%r**	**4%p, 8%r**
Restoration multiplier (range)	9.3–60	81.4–3'445	81.4–3'445	7'096–1.3E07
Averted loss multiplier (single value)	2.7	2.7	7.7	59.3
Median gain [WR_{marg}], CONS	4.6E-06	4.3E-06	1.6E-06	1.6E-07

Table 6.2 – Effect of changing the discount rate on offset ratio multipliers and discounted conservation value for the hypothetical compensation project. Discount rate (%) provided for restoration (r) and protection (p). Ratio multipliers calculated within the compensation project area for converted habitat (restoration) and existing natural habitat (protection or averted loss). Protection applied a single multiplier for all natural habitat. Median gain per cell (900 m resolution) shown across all pixels covered by the compensation project for the CONS scenario (only non-PA land considered additional).

For averted loss offsets, increasing the rate from 1% to 2% changed the multiplier from 2.7 to 7.7 across the project landscape, and median gains within the reserve system from 4.6E-06 to 1.6E-06, or a 75.3% decrease (Table 6.2). Thus the main determinant of our project gains (and compensation requirements and cost) was the discount rate applied to averted-loss offsets.

6.3.4 Compensation through equivalence in loss and gain

Area-based compensation. We quantified offset ratios for each land use class as the ratio of hectares of area offset to compensate one hectare of impacts (i.e. offset ratios). To do this, we compared median values for per-hectare impacts and discounted per-hectare gains. We used median values due to the non-normal distribution of the data (average impacts and gain values were strongly skewed by extremes and thus much higher than the median). Median offset ratios across land use classes and scenarios (CONS and PPA) for WR_{marg} ranged 0.6:1 to 27.8:1 for impacts at the regional scale (East African), 7.2:1 to 19.2:1 at the national scale (Kenya) and 7.2:1 to 18.3:1 at the sub-national scale (Appendix, Table A.8.1 and Section A.8.2). Estimated annual costs per hectare compensated across land use classes ranged \$50–\$2'110 (regional), \$680–\$1'190 (national) and \$680–\$1'830 (sub-national) for the "CONS" scenario, and \$120–\$4'850 (regional), \$1'560–\$3'340 (national) and \$1'560–\$4'200 (sub-national) for the "ALL" scenario (i.e. including both PA support and development funds). These annual costs imply compensating one year of agricultural production or urban development on 1 ha by funding conservation of an appropriate offset area for 1 year.

Product-based compensation. Using WR_{marg}, we allocated the respective annual costs per area to land use products (tonnes of crop equivalent and ha of urban land) using production and price information. This led to median predicted price increases ranging up to about 35% for the "CONS" scenario (only conservation gains on non-PA land considered) at the national (Kenya) scale of impact assessment (Fig. 6.8 and Table A.8.1 in the Appendix). This increased to almost 150% for the "ALL" scenario (including half the management costs of public PA cats 5 & 6 and socio-economic development funds). Upper (75%) quartiles ranged up to 50% and 140%, respectively. We only report the national scale results because these are likely to be most relevant to any foreseeable application (i.e. product origin, agricultural area, production and price data are globally available at the national scale via FAOSTAT, 2013b), although regional and sub-national results are presented in Appendix 8.5.5 (Figs. A.8.5 and A.8.6).

We calculated results using the alternative species weighing system (WR_{hist}) and using a log-normal distribution of impact and gain data for WR_{marg} (to account for skew). Predicted premiums were lower for both alternative analyses, roughly by a factor 2–5 results (see Appendix 8.5.5, Fig. A.8.4 for WR_{hist} and Fig. A.8.7 for log-normal data).

We also applied the compensation calculation to the land use impact assessment results of de Baan *et al.* (n.d.) for three specific crops in East Africa (tea, coffee and tobacco). Estimated price premiums of the crops ranged 22% to 90% for occupation impacts and 10% to 280% for transformation impacts (depending on whether median or upper quartile impacts considered; Table 6.3). We also investigated the annual conservation revenue that could be generated by compensating a share of the exported volumes of these crops (using shares of 5%, 10% and 20%). These shares roughly correspond to market shares of organic (ca. 5% in the EU; www.organic-world.net) and fairtrade produce (depends on the product, but up to, e.g., 20% for coffee in the U.K.; www.tradeforum.org). Our results indicate that combined compensation of occupation and embedded transformation impacts could generate annual conservation revenues of \$0.9–\$16.9 million per crop at a 5% share, \$1.9–\$33.9 million at a 10% share, and \$3.7–\$67.8 million at a 20% share (all figures expressed in 2010 dollars; Table 6.3).

6.4 Discussion

In this study, we prospectively assessed ecological and economic aspects of an international biodiversity compensation scheme based on evolving methods for designing biodiversity offsets. We presented a proposed conceptual framework and outlined application to a hypothetical case study of a large-scale conservation project in Kenya offsetting the biodiversity impacts of land use at multiple spatial scales. We used local threat and rarity weighted species richness of mammals as a biodiversity "currency" (de Baan *et al.* , n.d.) to quantify losses and gains in conservation value. We applied an empirical model of habitat recovery (Curran *et al.* , 2014) to adjust expected conservation

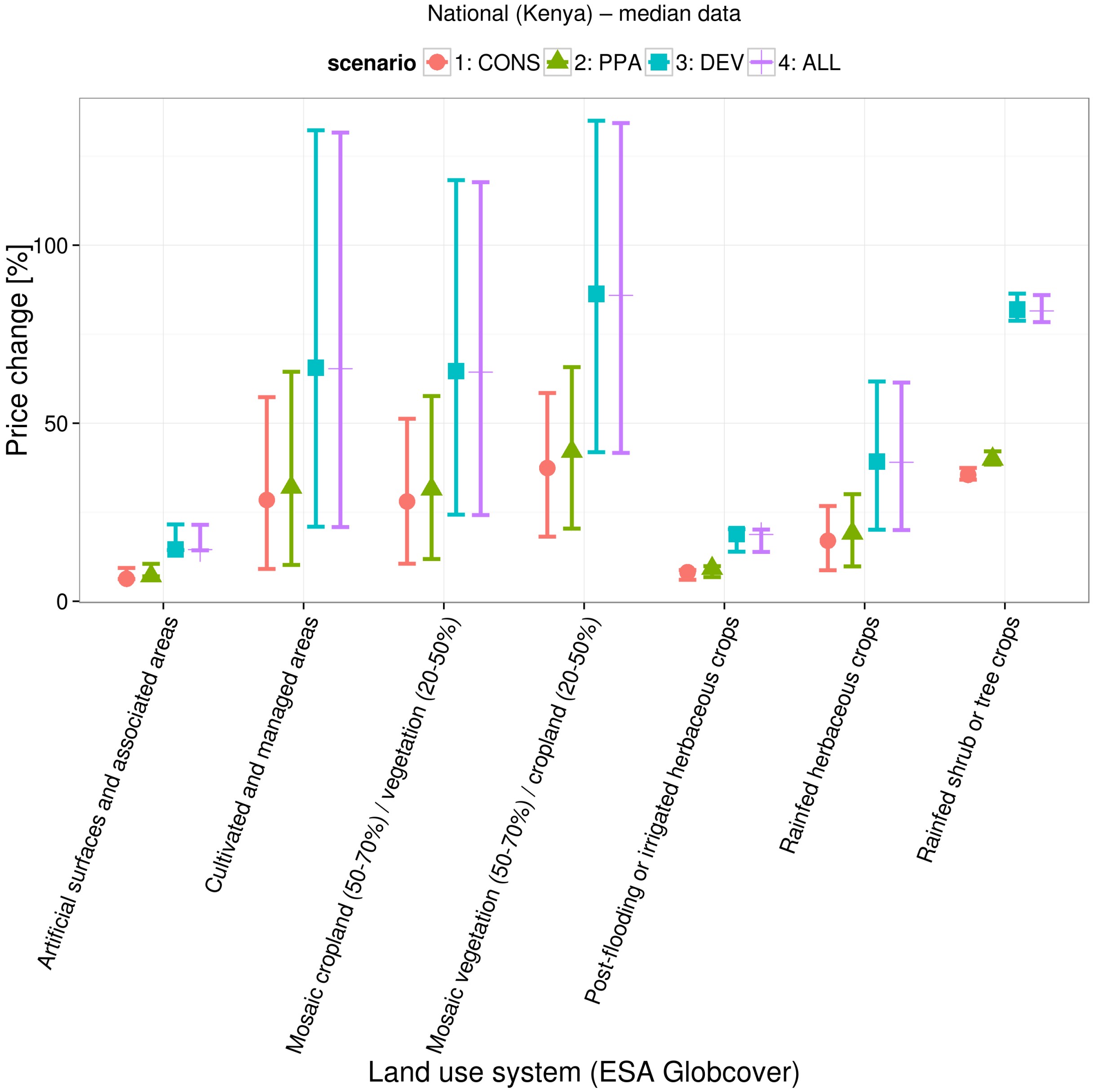

Figure 6.8 – National scale predicted farm-gate price premiums per land use product (tonne of crop equivalent, ha of development land) for ESA Globcover land cover classes using the *marginal* loss method (WR_{marg}). Results shown for four scenarios. Data range represents median and inner quartiles (25 and 75 percentiles). CONS = only non-PA land considered; PPA = support included for half the management costs of public PA cats 5 & 6; DEV = includes development budget; ALL = includes both development budget and public PA support.

Product	**Impact**	**Comp. area**	**Premium**	**Export val.**	**Revenue [mil. $/y]**		
	[*sp. eq.*/kg/y]	**[ha]**	**[%]**	**[M$]**	**5%**	**10%**	**20%**
Occupation impacts (land use)							
Tea, median	1.78E-10	3.34E-03	21.57	417.66	4.50	9.01	18.02
Tea, Q3	4.18E-10	7.83E-03	50.48	417.66	10.54	21.08	42.17
Coffee, median	5.71E-10	1.07E-02	31.76	44.29	0.70	1.41	2.81
Coffee, Q3	1.62E-09	3.03E-02	90.00	44.29	1.99	3.99	7.97
Tobacco, median	2.52E-10	4.73E-03	33.74	30.24	0.51	1.02	2.04
Tobacco, Q3	5.73E-10	1.07E-02	76.64	30.24	1.16	2.32	4.64
Transformation impacts (land use change)							
Tea, median	9.85E-09	9.23E-03	59.53	417.66	12.43	24.87	49.73
Tea, Q3	2.25E-08	2.11E-02	136.03	417.66	28.41	56.82	113.63
Coffee, median	3.56E-09	3.34E-03	9.91	44.29	0.22	0.44	0.88
Coffee, Q3	1.65E-08	1.55E-02	46.02	44.29	1.02	2.04	4.08
Tobacco, median	1.85E-08	1.74E-02	123.99	30.24	1.87	3.75	7.50
Tobacco, Q3	4.19E-08	3.93E-02	280.11	30.24	4.24	8.47	16.94

Table 6.3 – Compensation costs and potential conservation revenues for exports of three cash crops from Kenya (tea, coffee and tobacco). Compensation applies the $WR_{marginal}$ method of de Baan *et al.* (n.d.) for both occupation *and* transformation impacts (shown separately), using the conservation only (CONS) cost scenario. Crop-specific impacts were derived from taking spatial FAO production data for individual crops (from Monfreda *et al.* , 2008), and calculating impact statistics within this area. Crop impacts assessed based on median and upper quartile (75 percentile, Q3) for the East Africa study region. Conservation revenue assumes different shares (5%, 10 and 20%) of total crop exports are compensated (within the range of organic and Fairtrade market shares for various crops in the EU). Price data from FAOSTAT, 2013b. M = million.

gains from habitat restoration and to account for restoration failure risk. Using data on local conservation costs (Curran *et al.* , n.d.a), we simulated the compensation of land use impacts due to agriculture and urban development, predicting effects on generic product prices. We also applied the framework to three specific crops, estimating the potential revenues that could be generated for conservation provided a share (5%, 10% or 20%) of total exports would be compensated. Overall, our results suggest "biodiversity neutral" production could be achieved with farm-gate price premiums averaging 23% for the cheapest scenario, which focuses only on conservation outside of existing PAs (national average of "CONS" column in Table A.8.1 in the Appendix). Adding an integrated development dimension, along with financial support for public PAs, roughly doubled these costs to 81%. Addressing individual crops, and integrating embedded land use change impacts, led to higher estimates ranging 40%–160%, depending on the crop (combined median values for occupation and transformation in Table 6.3). However, our case-study results are context specific and relatively sensitive to the assumptions and methodological choices that we adopted. They are therefore primarily illustrative, helping to highlight important issues for debate and future research, some of which we discuss below.

6.4.1 Discounting, environmental value and biodiversity offsets

While significant progress has been made in resolving biodiversity measurement, delivery risk and uncertainty problems across spatial and temporal scales for offset calculations (Overton *et al.* , 2013; Moilanen *et al.* , 2009a; Pouzols *et al.* , 2012; Moilanen, 2012a), the choice of discount function and rate applied to temporary biodiversity debts (time-lags in delivery) remains an outstanding issue. Regardless of choice of rate, the practice is itself controversial because it assumes current losses and discounted, but as-yet unachieved, future gains are *qualitatively* equivalent (Overton *et al.* , 2013). This assumption is controversial given the potentially irreversible consequences (e.g. species extinction) of mismanagement (Bekessy *et al.* , 2010) and very poor success record of existing offsets and restoration projects (Curran *et al.* , 2014; Bull *et al.* , 2012; Maron *et al.* , 2012). However, Overton *et al.* (2013) highlight that restoration offsets are expanding in environmental policy globally despite concerns by ecologists (Maron *et al.* , 2012). At a minimum, a robust discounting and uncertainty framework can replace often arbitrary ratios with multiplier grounded in ecological theory and empirical data (e.g. Curran *et al.* , 2014), making blatantly unfair trades and net losses of biodiversity transparent to public scrutiny.

In our compensation framework, this issue is less important because we assume land use impacts are abstract, spatially vague, not preventable and largely historic (although we did address ongoing, embedded land use change or “transformation” impacts for selected crops; de Baan *et al.* , n.d.). In this context, we assumed temporally bounded impacts (i.e. per area and year) occur in the present, and developed a compensation framework based on funding an equivalent gain or maintenance of biodiversity for an identical time period, averaged over an assumed 100 y compensation project horizon. The long time horizon ensures permanence in conservation gains and is analogous to the minimum “crediting period” requirement for carbon offset projects of 30 y under “Gold Standard Certification”[2]. Due to the time profile of these gains versus the need for immediate compensation, an appropriate discounting framework is required.

Why and what to discount?

A first step towards choosing a rate is knowing why we should discount conservation value in the first place. Overton *et al.* (2013) provides five reasons drawn from service discounting in Habitat Equivalency Analysis (Dunford *et al.* , 2004) and utility discounting in economics (see Frederick *et al.* , 2002, for a review): (1) the risk of non-delivery of future conservation gains, (2) the “lost opportunity cost” of the use of biodiversity (i.e. the debt cannot be “used” by humans until it is recreated) (3) the “rate

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[2] http://www.goldstandard.org/

of return on biodiversity capital" (i.e. the reduced capacity for population reproduction, growth and speciation leading to reduced value *to humans* in the future), (4) changes in the marginal value of biodiversity to humans, (5) pure time preferences (i.e. individuals favour values now rather than in the future). These justifications for discounting are problematic because they do not clearly differentiate between *anthropocentric* (use and non-use) values and *ecocentric* (intrinsic) values (Chan *et al.* , 2012), nor define the role of offsets within these value systems. This lack of definition can also be found in the definitions of offsets used by practitioners, such as that of the Business and Biodiversity Offsets Programme (BBOP), which defines the goal of offsets as achieving a no net loss or net gain "with respect to *species composition, habitat structure, ecosystem function and people's use and cultural values* associated with biodiversity"[3] (emphasis added).

Whether to discount, and choosing the appropriate rate, will depend on the type of values being considered. The first three elements of the definition of offsets above seem to relate to ecocentric values, which we interpret as ensuring the persistence and diversity of affected biodiversity features, the resilience of ecosystems or other large-scale units, and preventing irreversible change. The presence of "no go" provisions for sensitive or vulnerable areas in offset policy generally reinforces this focus on ecocentric values (Pilgrim *et al.* , 2013).

The ecocentric value of biodiversity can be linked to economic rationality, but *not* via the neoclassical economic framework of subjective preferences revealed through markets. Rather, precautionary concepts from the (trans-)discipline of ecological economics (Farley, 2009), incorporate ecological value through capital substitution assumptions (e.g. strong sustainability) and concepts such as "critical natural capital" or the "safe minimum standard" (Berrens, 2001; Ekins *et al.* , 2003) Ecocentric values are also shared by approaches in conservation planning where rare, vulnerable or functionally-important biodiversity features often receive a higher priority weight. Thus, incorporation of benefit functions for marginal conservation value from spatial conservation planning is an important recent development (e.g. Pouzols *et al.* , 2012).

At the same time, anthropocentric values for biodiversity also make up a component of the BBOP definition of offsets ("*...people's use and cultural values* associated with biodiversity"), and appear in similar definitions for other policies (e.g. US wetland banking, various "Green" offsets in Australian states). This component of value presents an almost insurmountable challenge, given the complexities of social value systems. Chan *et al.* (2012) recognize eight dimensions of values, ranging from market-mediated exchange values to "transformative" or "metaphysical" (spiritual) values. Because of the context-specific, incommensurable and intangible nature of these values, a range of methods from the social and natural sciences is required. Simple market-based tools,

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[3] http://bbop.forest-trends.org/

such as offsetting or payments for ecosystem services, are bound to underestimate the information costs and uncertainties at work, leading to what Norgaard (2010) refers to as a "complexity blinder". Thus in our framework and case study, we generally focused on ecocentric values reflected in the benefit function and resulting "currencies", and retain this focus when discussing each of the justifications for discounting of Overton *et al.* (2013).

Risk of non-payment. Within the framing of Overton *et al.* (2013), discounting ecocentric values can be linked to the first three justifications, i.e., non-delivery risk, "rate of return on biodiversity capital" and the debt-induced "opportunity cost". Discount rates for these components should be based on (objective?) ecological variables reflecting changes in the state of ecological health and diversity over time. For the risk of non-delivery, we used 1% for both types of offsets based on habitat loss rates in Kenya (see Methods 6.2.6). Because this is an ongoing risk that grows with over time, it is appropriate to represent it in the discounting term. However, a binary risk (such as for restoration failure) is a function of area or management regime or other such planning variable, and is best dealt using a robustness correction rather than discounting (Moilanen *et al.* , 2009a; Overton *et al.* , 2013). We applied a correction to restoration offsets to such a restoration failure risk based on empirical data from the restoration and recovery literature (Curran *et al.* , 2014).

Opportunity cost. The "opportunity cost" effect could be thought of as the direct debt/delay of ecological value caused by the offset. Since averted loss offsets protect already-existing conservation values, the rate is 0%. For restoration offsets, we used a middle value of 1.5% from the recommended range of 1–2% of Overton *et al.* (2013), although no explanation is given for this range. One alternative would be to base the rate on the proportion of total conservation value in a landscape that is impacted by a project (or in our case, the median impact value divided by total weighted richness in the landscape).

Rate of return. The "rate of return" component reflects the rate at which marginal units of biodiversity replenish and grow (including over evolutionary timescales). This might reflect the contribution of the impact site to the maintenance and growth of conservation value in the landscape (e.g. as a source for emigration to other areas). This is often reversed for restored areas, which act as sinks for many threatened species until they reach a late stage of succession, even then exhibiting reduced diversity across a range of indicators (Curran *et al.* , 2014; Gibson *et al.* , 2011). Deriving estimates for this marginal productivity effect of biodiversity will require further research integrating life-history information, meta-population dynamics *(e.g. Hanski, 2011),* evolutionary history (e.g. Jetz *et al.* , 2014) etc. In our case study, we simply used the middle estimate of 1.5% recommended (again, without specific grounds) by Overton *et al.* (2013).

In economic cost–benefit analysis, discounting of monetary values is often applied based on the rate of return on capital, approximated in interest rates (e.g. on short-term risk-free government bonds; Gowdy *et al.* , 2011). This assumes a principal of weak sustainability, where increased wealth can compensate for environmental costs shifted to the future (De Groot *et al.* , 2013; Gowdy *et al.* , 2011). In the U.S., the National Oceanic and Atmospheric Administration recommends a discount rate for Habitat Equivalency Analysis (HEA) close to the recommended interest rate for environmental valuation (Dunford *et al.* , 2004). This is problematic even from a weak sustainability perspective as estimates of wealth increases do not take into account environmental externalities or social costs (TEEB, 2008; Gowdy *et al.* , 2011). HEA also applies these discount rates, not to monetary ecosystem service estimates, but to ecosystem compositional and functional indicators, raising issues of comparability and relevance.

Changing marginal value. Regarding the fourth justification for discounting ("changing marginal value"), Overton *et al.* (2013) suggest incorporating this into the measure of marginal conservation value rather than include it in the discount rate (otherwise discounting at 0%). We investigated using an alternative index that indicated historic effects of biodiversity loss and the increased value of scarcer biodiversity features (analyses with WR_{hist}). In this case, our estimates of impacts/compensation requirements were generally lower and we omitted a significant portion of "gains" in biodiversity from existing (near-)natural areas simply due to the increased scarcity of their component species. This scarcity effect on marginal value works like a negative discount rate, reducing penalties for time lags and uncertainties etc., and bridging between ecocentric and anthropocentric values (e.g. both Blignaut & Aronson (2008) and (De Groot *et al.* , 2013) recommend using a negative discount rate in economic valuations of ecosystem restoration and other projects of delayed benefit, based on increasing scarcity rents of ecosystem goods and services in the future).

Pure time preferences. Pure time preferences directly relate to anthropogenic values, and represent the outlook of individuals (individual discount rate) or society as a whole (social rate of time preferences). The latter is relevant to public policies such as offsets, and generally receive a low value in cost-benefit analyses of larger environmental issues such as climate change (Gowdy *et al.* , 2011; Gowdy, 2007). Overton *et al.* (2013) recommends a default zero value for this component. However, research into psychology and behavioural economics could illuminate useful concepts to adjust offset ratios (e.g.

widespread observations of loss-aversion show that people value what they have more than what they seek to gain, which perhaps has implications for setting offset ratios and multipliers).

Ratio multipliers and the feasibility of restoration offsets

Depending on the type of values, positive or negative discounting might be applicable, preferably using a range of rates (Gowdy *et al.* , 2011; De Groot *et al.* , 2013). Additionally, for some components of the discount rate, it is appropriate to discount the impact as well as the gain (e.g. habitat loss will affect the impact site, even if development does not occur, thus impact is discounted to reflect this reference scenario; see Fig. 6.3), and generally apply different rates based on the various components of the discount rate (Moilanen *et al.* , 2009a; Overton *et al.* , 2013).

Our sensitivity analysis illustrated a large influence of the choice of discount rate on estimated restoration ratio multipliers (Table 6.2). However, because our hypothetical compensation project was mainly composed of averted loss gains, changing the restoration discount rate did not significantly affect our estimates of median gains or our compensation estimates. We also showed that using time-averaged multipliers (i.e. accounting for an intermediate linear increase in biodiversity value in recovering habitats before they reach maturity) roughly reduced gains by a factor of 3–4 (Results, section 6.2.7). We employed ratios at maturity because it is unclear how our conservation currency (weighted richness) changes over time, which depends on the habitat preferences of the rare and threatened species. Our approach assumes only mature secondary habitat is of core value to such species (e.g. uGardner *et al.* , 2007a; Gibson *et al.* , 2011).

In general, our results indicate very high ratio multipliers for restoration offsets. Even discounted at lower rates, or using time-averaged multipliers, ratios typically ranged into the thousands, with mean values in the tens to hundreds in our case study region. Multipliers were even higher for other parts of East Africa (particularly montane areas) not shown in our results. Thus, we paint a rather pessimistic picture of the potential for restoration offsets to compensate losses of existing habitat (see also Curran *et al.* , 2014; Moilanen *et al.* , 2009a). However, as illustrated above, more research is required to elucidate appropriate discount rates under various scenarios. The exponential behaviour of our currency (reflecting inverse species range size and threat) means that across the analysis region, impacts and expected gains differed by orders of magnitude (Fig. 6.4). Hotspot areas covered the potential ranges of a number of critically threatened species, including the Giant Thicket Rat, *Grammomys gigas* (Dollman 1911), the Aberdare Shrew, *Surdisorex norae* Thomas 1906, the Smoky White-toothed Shrew, *Crocidura fumosa* Thomas, 1904, the Ultimate Shrew, *Crocidura ultima* Dollman 1915, and the

East African Highland Shrew, *Crocidura allex* Osgood 1910. Therefore, despite very large multipliers (i.e. ranging hundreds or thousands), restoration gains might still be worthwhile if expected marginal biodiversity value is high enough. However, this requires further studies of potential case studies in different world regions, and using different currencies.

6.4.2 Economic and institutional considerations

Product application and willingness to pay. Our estimated price premiums represent ballpark estimates of possible implementation costs, ranging tens to over one hundred percent of farm-gate prices (inner quartiles of estimates). Costs to the final consumer in the North will be lower due to the low relative importance of production costs on final consumer prices in North–South supply chains (Kaplinsky, 2000; Talbot, 1997; Bacon, 2005; Fitter & Kaplinksy, 2001). Accounting for this effect implies premiums about 5 or 10 times lower in the Global North (i.e. upper quartiles ranging up to 10% for the “CONS”, and 28% for the “DEV” scenarios).

These figures are optimistic, as they omit important transaction costs. In our hypothetical case study project, we considered some national transaction costs for conservation in Kenya Curran *et al.* (based on n.d.a), including administration, information, negotiation and monitoring costs. However, we did not include international transaction costs, which are likely to be substantial. Regarding International Payments for Ecosystem Services (IPES), Farley *et al.* (2010) recommends adapting existing global institutions to prevent new transaction costs from emerging from new conservation instruments. The GEF and CBD in particular stand out as potential administering, regulating and clearing-house bodies for such cross-border payments.

We presented crop-specific compensation data to give a sense of magnitude to the economic effect of compensating a fraction of exports (Table 6.3). With 5% of exports compensated from each crop, annual revenues might range \$1–\$17 million (in 2010 dollars). 5% roughly corresponds to the area share of organic agriculture in the EU (www.organic-world.net), which could be a suitable reference for market uptake of a voluntary offset scheme. In comparison, the 2007/2008 total annual budget for the Kenya Wildlife Service, which oversees the protection of wildlife within Kenya’s national parks, was \$54 million (also expressed in 2010 dollars; Bird & Kirira, 2009). Annual funding generated by the CBD LifeWeb Initiative to cover global project implementation costs was \$250 million in 2011. Thus our figures, if correct, may be quite substantial in raising funds for conservation. But we caution that these uncertain results should be interpreted with care.

Judging by consumer willingness-to-pay (WTP) studies for environmentally certified products in the North, our estimated price increases are modest. In the only specific WTP study for biodiversity friendly products, Bateman *et al.* (2010) finds WTP values for hypothetical "tiger-friendly" palm oil of 36%–56% in England. Other ethical and organic labels have been more extensively studied, with average consumer WTP values reported in the literature ranging 46% (Fairtrade chocolate in France; Didier & Lucie, 2008), 10% (Fairtrade coffee in Belgium; De Pelsmacker *et al.* , 2005), 6.46% (Fairtrade coffee in the U.S.; Loureiro & Lotade, 2005), 39% (organic coffee in France; Didier & Lucie, 2008), 3.2% (organic coffee in the U.S.; Loureiro & Lotade, 2005) and 6.44% (shade coffee in the U.S.; Loureiro & Lotade, 2005).

Institutional constraints. We did not specify whether such an offseting scheme should operate under a mandatory (e.g. through international agreements) or voluntary (e.g. certified "compensated" products) framework. However, existing trade laws largely rule out mandatory implementation. Under World Trade Organization (WTO) rules, restrictions on trade may be justified based on *General Exceptions* stipulated in Article XX (WTO, 1947). Grounds for regulating trade include "the protection of human, animal or plant life or health" (paragraph (b)) and "the conservation of exhaustible natural resources" (including biological resources at the brink of extinction; paragraph (g)). However, a connection to domestic environmental policy must be established by the nation applying for the exception, and only taxes and duties are permitted, not quantitative restrictions in trade volumes (WTO, 1947). In the only precedence case relating explicitly to biodiversity loss, the U.S. successfully obtained an exception for shrimp imports from some S.E. Asian countries (due to the impact of shrimp trawling on wild turtle populations). The U.S. proved a link to domestic environmental policy by highlighting the migration of turtles between S.E. Asian and U.S. waters, thereby falling under the U.S. Endangered Species Act of 1973. However, the case was subsequently rejected by the WTO because of preferential treatment to selected countries, and the use of quantitative restrictions (trade bans) rather than taxes[4].

Implementing the compensation framework proposed here as an ecological tax on traded commodities would, *in theory*, be allowed under WTO rules, but only if the currency is strongly linked to domestic species extinctions in the producing country via the benefit function (required by paragraph (b) of Article XX WTO, 1947). Such a scheme would only be feasible at the point of export from nations of the Global South (on grounds of protecting domestic biodiversity). A relevant question is whether perceptions of the value of biodiversity in the Global South are strong enough to muster the political will to implement such a tax (most probably against the will of the North, which benefits

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[4] http://www.wto.org/english/tratop_e/envir_e/edis08_e.htm

greatly from ecologically unequal exchange). Price increases are likely to be perceived as an additional competitive disadvantage for Southern Countries already battling with deteriorating terms of trade for bulk primary exports. (Witness the successful challenge of the U.S. shrimp ban by the Southern nations affected, rather then the adoption of anti-turtle excluder devices in their trawling fleets). In this respect, a voluntary scheme at the level of Northern consumer or retailer, funding individual compensation projects, could be more appropriate as a starting point, at least until international trade policy matures to better facilitate environment-related protective measures.

6.4.3 Conclusions

We conducted exploratory research on the ecological and economic aspects of an international biodiversity offset framework, illustrating application to a case study of agricultural production in East Africa. While our framework and case study application was highly simplified and uncertain, it offers a workable starting point to investigate the larger role of offsets in international biodiversity conservation. In agreement with earlier research, initial findings are pessimistic regarding the likely feasibility of restoration offsets when empirical time lags for habitat restoration are used as a basis for discounting, although this is strongly dependent on the chosen rate. Future research should focus on strengthening the theoretical basis for discounting in offsets, and we made an effort to distinguish appropriate practices relating to anthropocentric and ecocentric values. In this vein, a deeper discussion of the role of offsets within human value systems is urgently required to accurately identify what exactly is being compensated. Finally, our case study revealed potential cost estimates that were reasonable given the results economic valuation studies of the environment. However, these are clearly context-specific and uncertain, with future research required to assess whether similar ranges are found for other world regions and contexts. Given the above, whether, and how, an international offset scheme could and should be implemented is still very much an open question.

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Chapter 7

Conclusions

7.1 Critical appraisal and scientific relevance of the thesis

THE dissertation investigated the potential to compensate the biodiversity impacts of land use due to agricultural production and international trade in the North-South context. The introduction (*Chapter 1*) briefly reviewed the ecological and economic foundations of biodiversity loss, describes the problem, the major drivers, the role of ecologically unequal exchange and the potential for addressing the issue using novel policy tools. The research focused on ecological compensation as a market-based tool to address ecological unequal exchange, because of its grounding in biophysical units of loss and gain, thus supporting strong sustainability (i.e. as opposed to monetized impacts). The tools and methods developed in the dissertation, culminating in the final compensation framework presented and applied in *Chapter 6,* improve our ability to assess and address impacts to biodiversity embedded in trade. However, until the approach is tested in different ecological and economic contexts, and important issues are resolved, such as data availability, appropriate multipliers, discounting rates and functions and the measurement of biodiversity value, the practical applications are likely to remain limited. In order to guide future research efforts, the following sections highlight progress made during the dissertation in terms of methods development, and contrast this to alternative approaches in the literature, and identifies outstanding issues to be addressed.

Impact assessment and the biodiversity "currency"

The assessment of biodiversity loss at the local, regional and global scale has been a major focus of ecological research for decades (e.g. Sala *et al.* , 2000; Kinzig & Harte, 2000; May *et al.* , 1995; Pereira *et al.* , 2010; Wright & Muller-Landau, 2006a; Xenopoulos *et al.* , 2005; MA, 2005c). Detailed tools exist in the ecology and conservation planning literature to quantify impacts at the genetic (Faith *et al.* , 2010a; Faith, 2002; Faith *et al.* , 2004b), species (Elith *et al.* , 2006; Austin, 2007; Thomas *et al.* , 2004; Visconti *et al.* , 2011), community (Ferrier *et al.* , 2007; Ferrier & Guisan, 2006; Gibson *et al.* , 2011; Sodhi *et al.* , 2009; Turak *et al.* , 2011) and ecosystem levels (Gibbs *et al.* , 2010; Koh & Ghazoul, 2010a,b; Gardner *et al.* , 2010; Wade *et al.* , 2003) in response to a range of pressures. Methods for linking such assessment to flows of commodities are also being developed (Shandra *et al.* , 2009a,b; Koh & Wilcove, 2008; Lenzen *et al.* , 2012). The key challenge of the dissertation is therefore not the development of new tools to assess biodiversity loss, but the selection of available data and methods to link processes at multiple scales in a consistent framework. In this respect, the issue of missing impact assessment tools does not imply a lack of methods, but rather the missing link between methods that can facilitate the analysis of international supply chains.

One key requirement in ecological compensation is the assessment of *equivalence* between of loss and gain of biodiversity (Moilanen *et al.* , 2009a; Bruggeman *et al.* , 2005; Parkes *et al.* , 2003). This requires a biodiversity "currency" that must be shared by impact assessment and conservation gain assessment. Assessment tools that operate on a large scale, such as the country level "biodiversity footprint" of Lenzen *et al.* (2012), are very useful for quantifying broad impacts associated to numerous drivers and products. They are, however, unlikely to function as an assessment method for quantifying loss and gain in a compensation framework. Localized benefits of conservation gains from existing or planned projects must be linked to less-detailed assessments of impacts from products of uncertain origin (i.e. local scale gains must be linked with losses assessed at the national or regional level). Country-level analyses of deforestation rates, species threats or other rough proxies of biodiversity loss can facilitate basic assessment of impacts, but are difficult to transfer to the local context.

For example, Lenzen *et al.*'s (2012) "biodiversity footprint" is particularly valuable as it covers all major drivers of biodiversity loss, including species invasion, habitat change, and overexploitation. The method uses a proxy of species threat similar to the IUCN species red list index (Bubb *et al.* , 2009b), which is also being developed for whole ecosystems (Keith *et al.* , 2013). However, the scale of analysis for such a coarse index is limited to large areas (Keith *et al.* , 2013; Bubb *et al.* , 2009b), because it relies on a simple aggregation of the presence and threat status of a species/ecosystem within a defined geographic area (Keith *et al.* , 2013; Bubb *et al.* , 2009b). This limits the usefulness as a currency to assess conservation gains of a compensation project. Likewise using averted deforestation as a currency represents another possible option (e.g. linked to a red list index for ecosystems), but the use of area-based currencies in existing offsets is heavily criticized because of perverse outcomes and inequitable trades. This largely results from not taking into account differences in the quality and conservation value of different habitat (e.g. ten Kate *et al.* , 2004; Maron *et al.* , 2012; Kihslinger, 2008). Hence recent proposals have suggested using very complex currencies to balance habitat trades in offset scenarios (Bruggeman *et al.* , 2005; Moilanen, 2012a; Moilanen *et al.* , 2009a; Bruggeman *et al.* , 2009; Pouzols *et al.* , 2012). Indeed, a detailed tool to assess offsets within a spatial conservation planning context, the *RobOff* software has recently been developed (Pouzols & Moilanen, 2013). This includes dynamic modelling of biodiversity change over time, the response of biodiversity response to pressures and conservation interventions, the integration of measures of marginal biodiversity value from spatial conservation planning and its analysis over space (the latter aspect required exchanging input and output data with existing conservation prioritization software such as Zonation; Moilanen *et al.* , 2009c). However, these complex latter approaches are computationally intensive and designed for the assessment of local offsets at a relatively small (landscape) scale. In the context of an international compensation scheme, a balance is invariably required between complexity and practicality to link large scale loss with local or landscape scale gains.

The dissertation (*Chapter 2*) thus proposes to expand the Life Cycle Assessment (LCA) framework to better utilize existing tools and data to assess biodiversity loss (Curran *et al.* , 2011). The bottom-up nature of (product-based) LCA facilitates detailed analyses of complex processes, offering a good basis for tackling such multi-regional issues. However, regionalization in life cycle impact assessment is a recent development (e.g. Pfister *et al.* , 2009) and work is required to develop the inventory flows and regionalized characterization factors to support such a spatially enabled LCA. This computational structure of LCA also provides challenges for biodiversity assessment due to the strong link to a functional unit, and assumption of a linear damage function (Udo de Haes, 2006). This raises problems primarily related to the non-linear response of biodiversity to disturbance (Udo de Haes, 2006; Geyer *et al.* , 2010b; Curran *et al.* , 2011). Other assessment methods in ecology take a top-down perspective to explicitly account for these non-linear effects (e.g. linking macro-level changes in habitat area to the number of species persisting in a region via the species-area relationship; Koh & Ghazoul, 2010a). Recent work in LCA has incorporated some of these concerns by linking large-scale models of ecosystem and species distributional change to inventory data on products and services for land use (e.g. de Baan *et al.* , 2013b,a; Coelho & Michelsen, 2014; Michelsen, 2008; Michelsen *et al.* , 2014), water use (e.g. Verones *et al.* , 2013a; Pfister *et al.* , 2009) and pollution impacts (e.g. van Zelm *et al.* , 2007a).

Chapter 5 of the dissertation represents a co-authored work that illustrates how the LCA approach can be expanded in this way. The chapter uses large-scale (globally available) habitat suitability models (Rondinini *et al.* , 2011a) for precise biodiversity assessment.

The study links impacts to modelled crop production data using freely available FAO data (Monfreda *et al.* , 2008). While illustrating an effective way of reduce inventory uncertainty, it comes at the expense of temporal precision (the data is valid only for the year 2000; Monfreda *et al.* , 2008). To address this temporal mismatch, *Chapter 6* uses countrywide FAO data (FAOSTAT, 2013b) to develop models of average crop production for different land cover classes. While the data is more recent, information on the exact crop is lost as a trade-off. *Chapters* 5 and *6* illustrate how impact assessment methods in LCA can be developed for suitable use in a compensation assessment. This is due both to the spatial precision of the approach (i.e. gains can be measured at the landscape or project scale) and the high conservation relevance of the unit of biodiversity change. Weighted species richness is commonly employed as a measure of site conservation value in conservation planning (Moilanen, 2007; Moilanen *et al.* , 2009b). Similar measures have been used in an offsetting framework to balance losses and gains (Moilanen, 2012a; Pouzols *et al.* , 2012).

Spatial scale of compensation, no-go areas

The global scale nature of the data required for the assessment in *Chapter 5* facilitates a geographically wide application. This could facilitate the assessment of compensation payments for international value chains involving impacts at different world locations. However, aggregation of impacts across a large spatial context is likely to lead to problems due to assumed substitutability between rare species in the assessment method (i.e. conservation gains for a rare species in one location can be used to compensate losses to another rare species elsewhere). Dealing with this issue in terms of determining appropriate scales at which impacts can be aggregated and compensated represents an important focus for future work. As discussed in *Chapter 6*, additional techniques from ecology, such as mapping fine-scale patterns in community similarity (Ferrier *et al.* , 2007; Kreft & Jetz, 2010), aggregating characteristic assemblages of species in discreet Ecoregion maps (Olson *et al.* , 2001; Overton *et al.* , 2009), or the use of large-scale mapping of "bioregions" (Kreft & Jetz, 2010) represent different options to represent community overlap in impact assessment. Recent guidelines and recommendations for implementing land use assessment in LCA (Koellner *et al.* , 2013b; Koellner & Geyer, 2013; Koellner *et al.* , 2012) currently propose regionalization (i.e. differentiation of impacts or "characterization factors") at the Ecoregion level, if possible. Such guidelines could also be adapted to the compensation framework by stipulating a geographic limit to compensation (i.e. impacts can be compensated within the same Ecoregion, biome or realm; Olson *et al.* , 2001). However, this requires further research, including inputs from work on "out-of-kind" offsets in the compensation literature, specifically by integrating complementarity information into the biodiversity benefit function used to derive the biodiversity "currency" (Pouzols *et al.* , 2012; Overton *et al.* , 2013).

Chapter 6 also illustrates the problem of basing the biodiversity benefit function heavily on species rarity and threat. The huge differences in impact between areas of high and low endemism illustrate the disproportionate bias exhibited by a number of rare species in the region. While the method consistently weighs global extinction risk, the exact weighing of different attributes (rarity, threat, etc.) requires further justification. This also raises the issue of "no-go" areas for offset design (ten Kate *et al.* , 2004), and the prospect of compensating past irreversible impacts. The very high upper values of the impact ranges for some land use systems documented in *Chapter 6* indicate essentially irreversible impacts to species that are critically endangered (i.e. where the area of cultivation lies within the range of a species with almost no suitable habitat remaining). This leads to very high occupation impacts, which were largely ignored for the analysis by taking median data. However a consistent rule of way of dealing with such impacts should be developed. Compensating such an impact would require unrealistic offset ratio multipliers (leading to thousands of ha offset to impacted), raising the possibility of setting an upper limit to the severity of impacts that can be compensated. Existing research has proposed mapping "no-go" areas based on patterns in species rarity and conservation priorities (Blundell & Burkey, 2007). Of particular interest is the network of critical sites of imminent extinctions ("Alliance for Zero Extinction" Ricketts *et al.* , 2005).

Another source of high ratio multipliers indicating irreversible (permanent) impact from the recovery time predictions from the model developed in *Chapter 3.* The model shows a large amount of uncertainty in predicting diversity response to restoration. However, the ability to estimate time lags and restoration failure risk using empirical data supports the establishment of ecologically fair offsets. In the impact and compensation models of *Chapter* 5 and 6, the resulting transformation impacts and offset ratios for areas with very long predicted recovery times (hundreds to thousands or years) also implies a degree of irreversibility that should be considered in both future land use assessments and compensation policy, similar to the treatment of rare species data above. In the meantime, recent literature should be periodically included in the model to improve its statistical power and reduce uncertainties is estimated parameters. This will improve its usefulness in guiding offset policy.

Finally, the dissertation only briefly considered the issue of project implementation and evaluation in the final compensation section (*Chapter 6*), although *Chapter 4* presented a more comprehensive assessment of cost and ecological effectiveness. While the chapter discussed the importance of considering social fairness and institutional requirements for different conservation strategies, a more detailed and comprehensive evaluation of any specific project would be required in an implemented scheme. In this regard, much literature exists from existing PES and literature from which to draw tools and concepts for measuring and assessing important properties such as additionality, leakage, equity and permanence within a conservation context at multiple scales (e.g. van Oosterzee *et al.* , 2012; Van Hecken & Bastiaensen, 2010; Pascual *et al.* , 2010; McKenney & Kiesecker, 2010; Schneider, 2009; Wuenscher & Engel, 2012).

7.2 Practical relevance of the research

The thesis contributes a number of concepts and tools for application in applied conservation and the management of natural resources. Of primary value is the compensation framework developed in *Chapter 6,* which represents the culmination of the research. This framework and other components of the dissertation are discussed in relation to achieving global conservation targets under the Convention on Biological Diversity. Among the most recent 2011–2020 *Aichi* targets of the CBD[1], the following five targets stand out as particularly relevant to the thesis:

- Target 1: awareness of biodiversity is increased in society, individuals are empowered to take action
- Target 2: harmful subsidies are removed, positive incentives are established
- Target 5: the loss of natural habitats is halted or reduced by at least 1/2
- Target 11: protected areas are improved and expanded (to 17% of the terrestrial area)
- Target 12: extinctions are prevented
- Target 15: 15% of degraded terrestrial ecosystems are restored
- Target 20: financial resources are increased from all sources

The compensation framework developed in the thesis could be used to take advantage of the willingness to pay of individuals through a voluntary offsetting scheme for agricultural products. Implemented either at the firm (e.g. supplier, retailer) or consumer level, this would contribute to both targets 1 (empower individuals to act) and 20 (increase financial resources), while additionally mitigating the rate of loss of diversity (target 5). Simply including better reporting of the biodiversity impacts of product (e.g. using the assessment framework presented in *Chapter 5*) would facilitate awareness raising among society at large, and help to better represent biodiversity loss in socio-economic choices.

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[1] http://www.cbd.int/sp/targets/

Compensation mechanisms also employ a combination of positive and negative incentives to motivate actors to change patterns in production and consumption (TEEB, 2011). In the case of the framework proposed in this dissertation, price increases caused by compensation payments (whether voluntary or compulsory) create a negative incentive to consume high impact commodities (Peterson *et al.* , 2008). At the same time, residual payments for embodied impacts increase the flow of funds to the Global South, providing a positive incentive to supply biodiversity (e.g. by private landowners or communities engaged in a PES scheme, such as that investigated in *Chapter 4*), thus supporting target 2.

If compensation payments were targeted through a prioritization step, as suggested in *Chapter 6*, this would directly reduce the risk of extinction by targeting payments to areas most important for retaining biodiversity (target 12). Depending on the model of implementation, extending support for existing protected areas (e.g. "PPA" support scenario in *Chapter 6*) as part of compensation activities would directly support target 11. The cost-effectiveness assessment presented *Chapter 4* also supports the improvement and expansion of protected area networks in Kenya (target 11), with the analysis particularly relevant to conservation organizations making on decisions where and how to use conservation resources most effectively.

Finally, recovery model developed in *Chapter 3* represents a valuable tool to support the use of ecologically fair offsets, and in guiding offset policy. It represents the first such generalizable empirical model to assess time lags and restoration risks. Greater application of the model could offer a better basis by which to negotiate offset transactions, which have until now been based on a form of educated barter that leads to a net loss of biodiversity (Walker *et al.* , 2009b). The model could be integrated into existing tools for assessing offsets, such as *RobOff* (Pouzols & Moilanen, 2013), to help conservation interest groups (e.g. Civil Society Organizations, community groups, conservation NGOs) interacting with entities employing offsets to judge when a trade is equitable, or at a minimum illustrate blatantly unfair trades (Overton *et al.* , 2013). A simple, ecologically sound basis for estimating offset ratios could provide enhanced bargaining power to those negatively impacted by development projects to demand more substantial compensation for losses.

Implementation options

Achieving effective conservation globally within current protected areas (PA) will require roughly doubling available funds for marine ecosystems (Balmford *et al.* , 2004) and tripling funds for terrestrial ecosystems (James *et al.* , 2001a; Balmford *et al.* , 2003a). A future-orientated expansion of the global terrestrial PA system to fill current gaps in species and ecosystem coverage is likely to require about a 10-fold increase in conservation funding (relative to current levels; Bruner *et al.* , 2004; McCarthy *et al.* , 2012). Additional investments will be required to adapt the PA network in response to climate change-induced shifts in species and ecosystem distributions (e.g. Hagerman *et al.* , 2010). The lions share of additional financing is required in developing (tropical) countries, where biodiversity is disproportionately concentrated and funding shortfalls most severe (Balmford *et al.* , 2003a).

Funding shortfalls originate both from the ineffective *collection* of required funds from those willing or mandated to pay (inadequate conservation financing), and the failure to *disburse* funds in a cost-effective manner (ineffective use of conservation instruments; Farley *et al.* , 2010). The disbursement issue has become a prominent focus of research in both spatial conservation planning (e.g. Naidoo *et al.* , 2006; Didier *et al.* , 2009; Curran *et al.* , n.d.a) and conservation strategy choice (e.g. Ferraro & Pattanayak, 2006; Ferraro & Kiss, 2002; Wunder, 2007). The collection of funds has received less systematic attention in the peer-reviewed literature (although see Balmford & Whitten, 2003; Farley & Costanza, 2010; Farley *et al.* , 2010). The thesis, and particularly the final contribution (*Chapter 6*) presents one way to shift the costs of compensation to the polluter via compensation payments. There are a range of options by which such a scheme could be implemented, both regarding the collection and disbursement of funds.

Collection

Tables 7.1 presents an overview of collection tools, both widely employed and novel in regards to current policy. Ecological compensation is clearly a polluter-pays approach, but could be adjusted to link to similar tools such as ecological taxation (i.e. the compensation costs are included as a tax, collected and disbursed by a central authority either to public or private biodiversity "providers"). On the other hand, a voluntary compensation scheme would tend more towards international markets for biodiversity (e.g. consumers or firms purchase "biodiversity credits" equal to their estimated impacts on some form of international biodiversity market), or indeed could be implemented through "green" markets through some kind of certification scheme for "biodiversity neutral" products. In any case, there are very important ethical, social and economic (and likely ecological) implications for how the scheme could be designed, including the equity and fairness of the payments (e.g. a voluntary variants would capture willingness to pay whereas a tax might affect some social groups regressively) and the reliability and amount of payments (e.g. a general tax would yield a much more regular stream of income whereas markets suffer from volatility).

Financing mechanisms	Principle	Description
Multilateral transfers	Beneficiary pays	International financial transfers linked to aid (e.g. official development assistance), debt relief (e.g. Debt-for-Nature swaps) or cross-border positive externalities (e.g. the GEF, REDD, CDM). Generally such transfers are administered through national or international funds which are filled through voluntary contributions by governments or via bilateral/multilateral agreements (Oliver, 2004a).
Ecological taxation	Polluter pays	Environment related taxes placed on the consumption of goods and services, or financial services (e.g. Tobin tax). If taxes are applied in proportion to the biodiversity impact of a good or service, mechanism will apply the polluter pays principle. Farley *et al.* , 2010 suggest an international value added tax modelled on the "ICMS Ecológico" of certain Brazilian states (see Ring, 2008).
Ecological compensation	Polluter pays	Compensatory action (mitigation) taken to offset residual ecological impacts of a development project, often aiming for a "no net loss" or "net gain" of biodiversity or other ecosystem services over the project planning horizon (e.g. biodiversity offsets). In many countries, compensation is a legal requirement for activities damaging particular species or habitats (e.g. U.S. Endangered Species Banking, the E.U. Natura 2000 network). Some international firms integrate voluntary offsets within their environmental policy (TEEB, 2011, ch. 7), and an framework for mandatory international offsets on high-impact commodities has been proposed by Peterson *et al.* (2011).
Ecotourism and user fees	Beneficiary pays	Cross-border tourism largely driven by the private sector, but strongly supported by public infrastructure and partly funding many public PAs.
International markets for ES	Beneficiary pays	The marketing of well-defined ecological services, or land uses likely to secure them, to potential buyers at the global scale (e.g. Ecosystem Marketplace, Green Development Initiative; see Wuenscher & Engel, 2012).
"Green" markets	Beneficiary pays	The marketing of "sustainably produced" or "biodiversity friendly" products such as shade coffee (Perfecto *et al.* , 2005). Generally applied to the preservation of agro-biodiversity and products collected from wild ecosystems (linked to disbursement tool of certification in Table 7.2).

Table 7.1 – Overview of selected conservation financing (collection) mechanisms relevant to international conservation efforts. Mechanisms classified according to beneficiary pays or polluter pays principle. The table focus on international financing mechanisms, omitting purely local or national proposals (based on Bayon *et al.* , 2000; Gutman & Davidson, 2007).

Disbursement

Regarding the disbursement of funds (Table 7.2, different strategies to implement conservation on the ground also imply very different social and economic effects, and are constrained or facilitated by local, national and global policies and institutions (e.g. Pascual *et al.* , 2010; Proctor *et al.* , 2008; Kosoy & Corbera, 2010; Fairhead *et al.* , 2012; Norgaard, 2010; Curran *et al.* , n.d.a). Table 7.2 highlights a range of conservation instruments for implementing biodiversity policy, including both "command and control" (regulatory) approaches and market instruments (adapted from classifications of Hanley *et al.* , 2012; Miteva *et al.* , 2012; Pascual & Perrings, 2007; Engel *et al.* , 2008). Market

Conservation instrument	**Principle**	**Description**
Public biological resources		
National protected areas	Command & control	National protected areas established on public lands/waters, administered by government (Gaston *et al.* , 2008).
Decentralization measures	Command & control, property rights	Devolution of resource management rights to smaller political units/actors, from federal PAs to participatory community-based natural resource management (Miteva *et al.* , 2012).
Private or community-owned biological resources		
Environmental regulation	Command & control	Legal instruments to regulate the use of, or impacts to, biodiversity on private property or by private actors, usually relating to specific sites, species or habitat types (e.g., the US Endangered Species Act, CITES).
Subsidies, tax breaks and price controls	Positive incentive	Price subsidies, controls and tax breaks for biodiversity friendly agricultural production. While current agricultural subsidies/taxes/controls generally favour exploitation of biodiversity (and other ecosystem services), institutions could be restructured to favour agricultural biodiversity conservation (e.g. OECD agricultural subsidies to support the traditional farming of marginal lands; Lehmann *et al.* , 2011).
Integrated Conservation and Development Projects (ICDP)	Positive incentive	Combined conservation and development projects that subsidize capital inputs (e.g. equipment, capacity building) for biodiversity friendly enterprises (e.g., ecotourism, butterfly farming; Salafsky, 2011).
Product certification	Positive incentive	Certification of a production system as biodiversity friendly or ecologically sustainable (e.g. Shade coffee, Rainforest Alliance and Forest Stewardship Council; Ferraro *et al.* , 2005).
Landscape certification	Positive incentive	Landscape-level certification meeting broader conservation goals, but adding flexibility in how goals are met among regional actors (e.g. UNESCO Biosphere Reserves, the CBD Green Development Initiative; Ghazoul *et al.* , 2009).
Compensation type PES	No incentive	Compensation for opportunity costs incurred through the protection of biodiversity, often as a result of environmental regulation (e.g. compensation for forgone “potentially damaging operations” on Sites of Special Scientific Interest in the UK Wildlife and Countryside Act; Pascual & Perrings, 2007).
Reward type PES	Positive incentive	Voluntary transactions, not necessarily financial, in the form of rewards for a well-defined ecological services, or land use likely to secure it (e.g. Rewarding Upland Poor for Environmental Services in Indonesia, Nepal and the Philippines; Wunder & Albán, 2008).
Conservation easements	Property rights	The voluntary transfer of some (or all) land use rights from a landowner to a beneficiary at an agreed-upon price. While the landowner retains the title deed, and can sell the land to a third party, the easement remains in effect until use rights are purchased back from the beneficiary. Involuntary easements may also be obtained through a judicial process, with the landowner receiving compensation appropriate to forgone benefits (e.g. the Land Trust Alliance in the U.S.A.; Fishburn *et al.* , 2009).
Land acquisition	Property rights	Land purchases by an individual or organization for the purpose of wildlife conservation (e.g. private conservancies in East Africa; Jones *et al.* , 2005).

Table 7.2 – Overview of selected conservation (dispersement) instruments relevant to international conservation efforts. Classified according to command & control, property rights or incentive measures, and split into instruments for public or private/community owned resources. (based on Hanley *et al.* , 2012; Miteva *et al.* , 2012; Pascual & Perrings, 2007; Engel *et al.* , 2008).

instruments are further subdivided into property rights and (positive/negative) incentive instruments (Gustafsson, 1998). Due to separate approaches, instruments are classified according to publicly and privately owned biological resources, although some overlap occurs in, e.g., participatory or joint forest management. From an economic perspective, transaction costs should ideally be minimized and cost-effectiveness criteria should be adopted in both the choice of conservation strategy (e.g. Ferraro & Kiss, 2002; Ferraro & Pattanayak, 2006) and the spatial targeting of interventions at all scales from local to global (e.g. Naidoo & Iwamura, 2007a; Naidoo *et al.* , 2006; Wuenscher & Engel, 2012). Chapter 4 investigated this issue in Central Kenya regarding two of the above strategies (land purcheses/easements and PES), and integrated institutional and fairness concerns, but much more work is needed to facilitate the right strategy choice for the correct conservation context, taking account of the local structure of property rights, governance bodies, social customs, the distribution of access and use rights, and the specific ecological properties of the type of biodiversity in question. Not paying enough attention to such issues can lead to unforseen consequences that impact both social and ecological systems (e.g. the "tragedy of enclosures" when western-style private property, necessary for PES are imposed on a region or culture where customary, informal use rights prevail; Fairhead *et al.* , 2012).

Institutional barriers

While voluntary implementation represents an immediate option, it would be limited in uptake and restricted to only a proportion of the market. Uptake of the concept in a policy making framework (e.g. via ecological taxes on traded goods from high-impact areas; Peterson *et al.* , 2008) would lead to presumably wider coverage of products and larger conservation benefits. One main barrier however could be the allegations of "environmental protectionism" by trading partners. The literature on ecologically unequal exchange testifies to a possible conflict between economic openness and environmental protection (e.g. Talberth & Bohara, 2006).

The WTO and General Agreement on Tariffs and Trade (GATT) contain stipulations for environmental exceptions to (Artcle XX) to the principle of non-discrimination (i.e. that "like" products between countries cannot differ in preferential treatment). To qualify for an exception under article XX, a country must demonstrate a conflict with a domestic stated health or environmental policy. If differences in "process and production methods" (PPM) between domestic and foreign supply infringe on the policy, then the exception can be granted. Threats to other species and crucial ecosystems are covered by article, but must be linked to domestic policy. In a precedence case in 1997 (WTO dispute 8, http://www.wto.org/), the US successfully won an exception in 1997 to regulate turtle-damaging shrimp imports from a number of Asian countries (i.e. wild shrimp caught without the use of turtle exclusion devices) The US argues that that turtles migrate to US waters, are part of the US fauna and are therefore covered under the US 1973 endangered species act. However, the case was subsequently dismissed upon appeal due to the use of import restrictions rather than tariffs by the US.

The compensation framework presented in the dissertation is likely to conflict with WTO trade rules unless countries can also object based on conflicts with international agreements (e.g. the CBD commitments) rather than simply domestic policy. For the immediate future, the framework is therefore better suited to voluntary application.

Outlook

The work developed during the course of the thesis will should be built on in the future to focus both on methodology and conceptual underpinning. One interesting step is the development of compensation methods for multiple drivers of biodiversity loss (e.g. land and water use impacts under a common compensation framework using impact assessment methods from Verones *et al.* 2013a; de Baan *et al.* 2013b, n.d.). One challenge in such an approach is the use of a similar benefit function and marginal unit of biodiversity value. Whether or not to aggregate impacts, and allow compensation to substitute between terrestrial and aquatic habitats is an open question that requires a sound theoretical framework. Impacts to different ecosystem types (aquatic and terrestrial) will likely need to be considered separately (i.e. through separate compensation effects). In this vein, extending the recovery model developed in *Chapter 3* to aquatic habitats using available literature (e.g. Jones & Schmitz, 2009) would contribute towards the development of appropriate multipliers for such habitats. Additionally, a conservation strategy choice framework, similar to that of Chapter 4, and prioritization approach for aquatic habitats (e.g Turak *et al.* , 2011) will be required.

A second should up-scale the case study approach to more world regions and ecosystem types to better assess the feasibility of the scheme. This would offer a chance to test how the concepts and methods (e.g. multipliers, discount rates) perform in different ecosystem types and how cost estimates differ through space and world region. Including more explicit project evaluation studies that expand upon the research of *Chapter 4* would be informative as to whether the framework can be adapted to different socio-cultural, policy and legal contexts. For a larger assessment, a global scale targeting scheme would be needed covering ecological, economic and also socio-political implementation information (what Wuenscher & Engel, 2012 refer to as the “risk of non-provisioning” of an ecosystem service). While examples of individual components of such a targeting approach exist in the literature (e.g. Eklund *et al.* , 2011; Wuenscher & Engel, 2012; Naidoo & Iwamura, 2007a; McCarthy *et al.* , 2012), a consistent targeting framework that includes the most important factors (e.g. conservation value, additionality, leakage, governance and political risk) has yet to be developed and tested (Wuenscher & Engel, 2012).

While the compensation framework developed in this dissertation requires further development, the framework stands as a basis investigating options for measuring and compensating the biodiversity impacts of land use. The work represents an important step with respect to conceptualizing issues such as “ecological debt”, ecologically unequal exchange and the polluter-pays principle. Such concepts offer a powerful way to conceptualize the biodiversity crisis, particularly from the perspective of actors in the Global South who are disproportionately affected by ecological degradation.

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Chapter 8

Appendices

8.1 Appendices for Chapter 2

8.1.1 Genetic indicators

Techniques to measure genetic diversity can be divided into those covering intraspecific (within species) variation, and those representing interspecific (between species) variation.

Intraspecific variation may be quantified by characterising the *discreet* allelic state of a population, or using *continuous* genetic traits (Hughes *et al.* , 2008). This involves the use of visible polymorphisms, which are phenotypically expressed by the organism to give a discreet indication of allelic state (e.g. Mendel's pea experiments); molecular markers, which separate the products of transcription, proteins, using electrophoresis to discreetly assign allelic state; and DNA markers, which involve DNA sequence data and may be discreet, or continuous (Conner & Hartl, 2004).

Discreet indicators include allelic diversity, an index of the number and frequency of alleles in the population; allelic richness, the number of alleles per locus; genotypic richness, number of genotypes in a population; heterozygosity, the proportion of loci that carry two alleles within a single diploid individual; mutational diversity and effective population size; nucleotide diversity, the number of nucleotides differing between two random individuals from a population; and the percentage of polymorphic loci (Hughes *et al.* , 2008). Continuous indicators include genetic variance, the variation in a trait among individuals of a single pedigree calculated using parent-offspring regression or genealogical information; the coefficient of genetic variance, expressing the variance in a trait in relation to the trait mean in the sample; and heritability, the ratio of genetic variance in the individual to total phenotypic variance in the population (Hughes *et al.* , 2008).

Intraspecific genetic variation can be extremely important in small or isolated populations, where random genetic drift and inbreeding can reduce fitness and adaptability. This increases susceptibility to stochastic variation in population size or changing environmental and biological conditions, increasing extinction risk. High intraspecific genetic diversity can increase ecosystem resistance to environmental or biological disturbance (e.g. Hughes & Stachowicz, 2004; Zhu *et al.* , 2000), and increase the rate of recovery after such disturbances (Reusch *et al.* , 2005). Although important for both the maintenance of higher level biodiversity (Hughes *et al.* , 2008), and in maintaining and provisioning ecosystem services (Hajjar *et al.* , 2008), implementation of monitoring genetic diversity has not been prioritized by the Convention on Biological Diversity, despite its mandate to do so (Laikre, 2010). Consequently, availability of data on intraspecific genetic diversity lags behind those of species and ecosystems, despite the availability of indicators.

Interspecific (between species) genetic diversity has been included in a number of biodiversity assessment frameworks via the use “phylogenetic indices”, constructed using molecular phylogenies of various taxa (Faith, 2002). In particular, phylogenetic diversity (PD) is an important metric of biodiversity because it reflects evolutionary history of a community, is tightly linked to endemism and species complementarity (Faith *et al.* , 2004b), and acts as a proxy for changes in ecosystem function and functional diversity (Cadotte *et al.* , 2008). It is defined as the sum of the branch lengths linking individual species within a community or sample (Faith *et al.* , 2004b). The presence of evolutionarily old or relictual species, or the distribution of species among many taxonomic groups, increases this value. The approach by Cadotte *et al.* (2008) might be particularly applicable to LCA because it requires only species information and widely available and extensive sequence data from GeneBank (National Center for Biotechnology Information; http://www.cnbe.nlm.nih.gov/). Such an approach could theoretically be retrospectively applied to existing methods in LCA (e.g. land use plant sampling data) to create PD characterization factors in LCA when identities of species are *known* from the original research. A synthesis of methods and approaches employed in molecular systematics to derive phylogenies is given in San Mauro and Agorreta (San Mauro & Agorreta, 2010).

8.1.2 Species-based indicators

Indicators that describe the trend or condition in single species or their attributes (e.g. abundance, occurrence, fitness, breeding rate, distribution) are considered in this article collectively as species-level indicators. This applies whenever the trend in a reduced set of species is considered a proxy for changes across all species. The advantage of taking a species-level approach is that information requirements are reduced relative to taking all species into consideration. Additionally, more information is present because each species’ response is assessed independently, and the results are often intuitive for conveying information (Lamb *et al.* , 2009). In contrast, community-level indicators, discussed below, only show trends in a sample from a biological community (e.g. species richness or Shannon-Wiener index), and information is reduced to a single numerical value for the sample. Community indices such as the Shannon-Weiner index may be difficult to interpret to non-biologists. The distinction, advantages and disadvantages of the two approaches are discussed in further detail in section 3, with examples of both types of approaches highlighted in practice.

Species-level methods include the focal species approach (Lambeck, 1997), which selects a few species to act as “umbrellas” because of their habitat requirements or sensitivity to a particular pressure. If the most sensitive and relevant species are used as indicators, then the thresholds for the majority of other species for the particular threat or pressure is assumed to be safely below acceptable thresholds. But this assumes that sufficient knowledge is present to determine which species act as suitable indicators for all species (Ozinga & Schaminée, 2005). In practice, the focal species approach is controversial (Carignan & Villard, 2002; Lindenmayer *et al.* , 2002; Caro, 2003). A multi-species' approach, that incorporates many species as indicators, is more reflective and useful for reflecting general trends across all species than the use of specific focal species (Roberge & Angelstam, 2004).

Species-level approaches also include "intactness indices" (Lamb *et al.* , 2009; Scholes & Biggs, 2005; Alkemade *et al.* , 2009; Faith *et al.* , 2008; Rouget *et al.* , 2006b). These describe changes in the population intactness of individual species with reference to some baseline or reference state (giving an index ranging from 0, complete population loss, to 1, population equal to the reference state). Intactness indices are generally averaged across a set of species within a taxonomic group to yield more general assumptions about changes in biodiversity (Lamb *et al.* , 2009). One assumption is that a reference state does indeed exist, and is often approximated by using historical monitoring data (Loh *et al.* , 2005) or pristine habitat (Scholes & Biggs, 2005; Alkemade *et al.* , 2009). This may be criticised as inferring a "nature constant" view of biodiversity that ignored inherent variation. But reference points are often chosen based on measurable and changeable areas along with measuring the population of interest. For example, taking populations *inside* natural protected areas as a reference for species intactness *outside* protected areas allows integration of the temporal variation in population size within reference habitat as long as monitoring persists in the reference area (e.g. Loh *et al.* , 2005). One criticism that holds more merit is that intactness focuses on *quantities of individuals or populations* of single species, rather than the *representation of the variety* of species that is more in line with the definition of biodiversity given by the CBD, and represented by quantifying the percentage of lost species for example (Faith *et al.* , 2008). This criticism can be levelled at any species-based approach, and methods must be designed to specifically remove the possibility of a widespread dominant species biasing or obscuring losses to more rare or sensitive species when individual species responses are averaged or combined in some way.

More recent methods applying at the level of the individual species include establishing a Habitat Suitability Index (HSI) which is used to model micro- or macro-distribution of species, i.e. species dispersion across habitat, or range across ecosystems respectively. HSI can be constructed using an *inductive* or *deductive* process (Corsi *et al.* , 2000). Inductive approaches use raw data on species occurrence or abundance and associated environmental parameters. Patterns within the data are extracted via multivariate statistics, and attributed to changes in particular variables of interest. Species distribution modelling, using complex statistical procedures, is developing very rapidly as a means of constructing an inductively derived HSI based on the climate envelope of the species (Rondinini & Boitani, 2006; Austin, 2007). A range of methods have been developed to model ranges based on occurrence data (Elith *et al.* , 2006). In LCA, the method to construct characterization factors for acidification inductively modelled species occurrence as a function of nitrogen loading (van Zelm *et al.* , 2007a). Nitrogen deposition was modelled across Europe and estimated Base Saturation (BS) levels were used to predict the probability of occurrence of 240 species of forest plant while statistically accounting for the effects of other environmental variables. Inductive approaches do not require a mechanism to be proposed (i.e. why nitrogen deposition causes a species to be absent) and lead to continuous suitability measures rather than suitability classes.

Deductive methods employ well-established empirical relationships or expert judgement to estimate the effect of changing environmental conditions on biodiversity. Meta-analysis may be used to establish standard effect sizes that are assumed to apply across unknown data ranges. Species sensitivities or habitat associations may be based on life history information and expert judgement, which often produces discreet suitability measures that require habitat to be split into vegetation classes. Species are either present or absent in habitat, or habitat is classed into grades of suitability. The BioScore tool (Louette *et al.* , 2010) takes this approach in joining cause and effect across a number of pressure categories and indicator species. Another example of a deductive approach is the process of expert interviews and workshops that generated the distribution ranges for all known amphibian species in the Global Amphibian Assessment of the IUCN. The Natural Capital Project's InVEST tool (Nelson *et al.* , 2009) uses deductive species-habitat associations to predict the effects of land use change on species probability of persistence in a landscape. A similar approach is under development in LCA to predict the effects of land use change on biodiversity (Geyer *et al.* , 2010a). The choice between inductive and deductive approaches is largely a trade-off between data-availability and needs of the study (Ricklefs, 2004).

The above species-level indicators generally focus on the compositional and structural attributes of biodiversity. Some functional aspects at a species level can be measured using genetic indicators. These include using genetic indicators to monitor inbreeding and outbreeding trends, genetic bottle-necks, metapopulation structure, effective population size etc. Aspects of life-history such as growth, fecundity, feeding, morphology, nesting, breeding etc., are beyond the scope of this review.

8.1.3 Community indicators

Community-level indicators, such as species richness, reduce a large amount of information across many species into a single numerical value. They describe the emergent properties that result when the ranges of individual species overlap. We define them here as characterizing and summarizing information across objective, non-biased samples from a locality or region. Such as quadrat sampling data of plant species in a meadow ecosystem, or the results of observational transects to monitor bird occurrence and abundance. This is usually results in samples that describe the number and relative abundances of species from a local community (or any other unit of measurement such as genera, family, functional group, guild etc.). These can be divided into univariate metrics that describe alpha diversity (at the level of a locality) and multivariate metrics that describe beta diversity (the similarity or turnover of species between localities). Gamma diversity is often considered to be the total diversity of a region encompassing numerous localities (Lamb *et al.* , 2009). Local extinctions and colonizations lead to variation in the micro-distribution or dispersion of species in a particular habitat (Noss, 1990). Single localities therefore may lack many species that are found elsewhere in the same habitat and are recorded in gamma diversity, the sum of all localities within a region. It must be stressed that the concept of the discreet, closed, ecological community of characteristic organisms has become essentially a non-concept in biodiversity research (Rahbek, 2005; Ricklefs, 2004). There are only overlapping ranges. Where community indicators differ from species indicators is that they are unbiased samples across species, not targeted samples from single species. However, species indicators may be used to derive community-level data via, for example, layering individual species' ranges to obtain estimates of species richness at a resolution equal to the modelling "grain" used to predict the species' ranges (e.g. Beck *et al.* , 2006a). However, this assumes all species are represented in the modelling approach, or that patterns in species richness of rare species is reflected in patterns of common species which are more accurately modelled.

Community indicators can be abundance- or occurrence-based depending on whether information on the relative abundance of species within a sample is considered. Abundance can be expressed in individuals, biomass, percentage cover, extent of occurrence etc. Species richness is the most simple and widely used index of community diversity because its units are easily conceptualized and widely published. As more weight is placed on the relative abundance of species within a sample, importance is shifted away from species richness to community "evenness" the antipode of species richness being measures of pure evenness of a community, and does not consider the number of species). Intermediate points between richness and evenness include the Shannon-Wiener index, which values each species by its relative abundance, and the Guini-Simpson index, which weighs each species to the square of its relative abundance (Hill, 1973). Beta diversity can also be differentiated into abundance-based and incidence-based and illustrate the ecological distance or similarity between local communities. Beta diversity patterns over a large spatial scale are less well studied than extrapolations of local species richness (Beck & Chey, 2007). Yet, beta diversity patterns are important regional indicators of both biological value, and pressures such as the extent of biotic homogenization due to habitat degradation or land use change (Koellner *et al.* , 2004).

At least two major problems affect biodiversity data availability and reliability. Firstly, a lack of standardized methodologies during surveys, and ongoing developments in sampling methodology, has lead to outdated information and a poor ability to compare species lists across sites. This generally results from different sampling techniques being used, therefore different sections of communities are sampled, and differences between localities may be an artefact rather than a true pattern. Secondly, the failure to account for undersampling of local communities has lead to unreliable species occurrence data across many regions (i.e. not detecting the full complement of species because of the "rare" element that requires extreme sampling effort to detect; (Southwood & Henderson, 2000)). In recognition of the second point, a large body of research has been devoted accounting for undersampling (e.g. by rarefaction; Hurlbert, 1971), or estimating the unseen species based on the structure of sampling data. This includes species richness estimators that either fit a statistical distribution to the sampling data (parametric methods) or use the proportion of rare and common species in a collection of samples to infer how many more species are likely to be detected with future sampling (non-parametric methods; Colwell & Coddington, 1994; Chao, 2005; Brose *et al.* , 2003). Applying estimators to available data may lead to more reliable values for species richness, but information on the identity of species is lost, which can be informative in considering effects on certain species groups (e.g. threatened species; Koellner & Scholz, 2008) or incorporating beta diversity (Koellner *et al.* , 2004). Estimators have traditionally been applied to estimate local community species richness, but large-scale regional applications have yielded promising results (e.g. Desmet & Cowling, 2004; Beck & Kitching, 2007).

Parallel to research in modelling individual species distributions, methods in mapping "emergent biodiversity" have been developed recently to use community data such as species richness, endemism, and turnover, rather than the climate envelopes of single-species and their distribution ranges (Faith & Walker, 1996; Ferrier *et al.* , 2004; Arponen *et al.* , 2008). Elith *et al.* (2006) provides a comparison of "species distribution" and "community modelling" approaches developed in recent years. Because mapping individual species ranges is data intensive (especially for rarely recorded species), using available checklists to create community indicators shortcuts the modelling process. However, as highlighted above, information on individual species is lost in the transition. However, community modelling is a promising approach towards mapping poorly studied regions where distribution data across all species is not readily available (Ferrier *et al.* , 2004). Spatial mapping of beta diversity patterns was used recently to estimate biodiversity loss due to land use change in Madagascar (Allnutt *et al.* , 2008). This goes beyond the non-spatial and deductive approach of the species area relationship, but is less data demanding that using individual species distribution ranges to infer extinction risk (e.g. (Jetz *et al.* , 2007)). A combination of spatial turnover in species from point data and continuous environmental variables was combined to spatially map species dissimilarity across Madagascar using general dissimilarity modelling (Ferrier *et al.* , 2007). This was used to estimate the percentage of species lost to land conversion between 1950 and 2000 (Allnutt *et al.* , 2008). Of significance is the ability to spatially predict past hotspots of endemism, turnover, and species richness on converted land based only on readily available environmental information (see "Environmental Diversity" or ED; Faith & Walker, 1996).

The above-mentioned indicators generally focus on the composition of local and regional communities, but say little about physical structure or ecological function. Structural indicators at the local scale describe the complexity of habitat. This includes vegetation density, vertical layering and spatial microdistribution of organisms or species, canopy intactness, physiognomy or growth form etc. (see Table 2.1 of the main text). Structural information may also be approximated using remote sensing and GIS at variable resolutions, which can be very useful in scaling from local community to ecosystem and landscape level (Foody, 2003; Cannon *et al.* , 2007). This includes canopy cover and structure, gap density, habitat heterogeneity and even the identity and composition of tree species based on their canopy dome signature (Foody & Cutler, 2006).

Using community species lists, classification of species by functional trait has been used to infer functional group diversity (FD) using life-history information and species lists (Petchey & Gaston, 2006). Combining species and community indicators, Thuiller *et al.* (2006) modelled the distribution of 122 European tree species under various climate change scenarios. Change in FD was related to changes in emergent patterns resulting from range reductions and expansions across species (Thuiller *et al.* , 2006). Continuous indicators of functional diversity can be generated using a "distance in functional trait space" approach between species of a sample or community (Petchey & Gaston, 2006). Trophic diversity, food chain length, and guilds are also used to define functional diversity. Elsewhere, functional diversity has been approximated using indicators of genetic diversity. Cadotte *et al.* (2008) found phylogenetic diversity (PD) to be a better metric and predictor of functional diversity, expressed as primary productivity in a local community, than either species richness or FD. GIS is also used to map functional ecosystem processes such as fire and disturbance regimes, canopy moisture and phenological change (Foody, 2003). The Normalized Difference Vegetation Index (NDVI) has been used extensively at a range of scales, for example to map productivity and moisture patterns for drought and hydrologic regime classification.

8.1.4 Species or community indicators?

Theoretically, the difference in results between species-level indicators and community-level indicators is likely to disappear as more data becomes available across a more species, eventually covering all species. But until it is possible to model the response of each species independently, and cover the full number of species in a region, there is always an assumption that the species which are *not assessed* react in the same way as species that *are assessed.* Recent approaches that monitor changes in biodiversity across large spatial or temporal scales utilize very large numbers of species as indicators and are more likely to reflect changes across all species. For example, Thomas *et al.* (2004) modelled the effect of climate change by assessing how the distribution range of over 1000 species would change under different climate scenarios. A community-level approach would estimate the number of species in a particular habitat area (e.g. Afrotropical rainforest vascular plants), model the change in area due to climate change, and then apply a relationship to calculate the estimated species loss, such as the species area relationship. This approach was used in the Millennium Ecosystem Assessment (Van Vuuren *et al.* , 2006). Similarly, Jetz *et al.* (2007) modelled the effect of climate change and habitat loss on bird species richness using a species-level approach. The study assessed how the two pressures would affect the range size of all known bird species using deductively mapped distribution ranges (from Orme *et al.* , 2006). They calculated the resulting extinction risk on a per species basis, averaging to a percentage loss of species. A community-level approach would delineate habitat borders of major ecosystems or biomes, and species richness through local and regional checklists. The effect of pressures in reducing the habitat area would then be related to expected species losses, e.g. via the species area relationship or through community modelling of spatial alpha, beta and endemism patters (e.g. Allnutt *et al.* , 2008).

8.1.5 Ecosystem and landscape indicators

At larger spatial scales, indicators that reflect the extent, condition, threat level and biological importance of ecosystems, landscapes or regions are termed “ecosystem indicators”. They may be based on nested combinations of landscape indicators of ecosystem coverage, community indicators from point localities, species indicators of distribution or sensitivity, or even indicators of genetic diversity. It is important to understand how local impacts will affect regional biodiversity. This may reflect the state of ecosystem-level biodiversity, such as in the amount of remaining natural habitat relative to ecological requirements (Fahrig, 2001; Olson & Dinerstein, 2002; Potapov *et al.* , 2008a), the size of habitat patches within the landscape (Swift & Hannon, 2010), the level of fragmentation and connectivity (Herzog *et al.* , 2001), ecosystem structural integrity and composition (Cannon *et al.* , 2007), or levels of endemism, turnover, species richness and phylogenetic diversity (Burgess *et al.* , 2006; Faith *et al.* , 2004b). Functional aspects can also be reflected at the ecosystem level either scaled upwards from finer indicators, or modelled independently from remote sensing data (e.g. NDVI and its applications; Foody, 2003).

The relationship between structural landscape *patterns* (i.e. how humans perceive landscape structure) and their effect on biological *processes* (i.e. how the pattern affects species and assemblages) remains poorly understood (Czajkowski *et al.* , 2009). The development of metrics of landscape pattern has for the most part stabilized, and the links to landscape processes has focused heavily on area loss and fragmentation (Fahrig, 2001; Turner, 2005; Swift & Hannon, 2010). Major metrics of pattern calculated by the FRAGSTAT program are classed into the following categories (Herzog *et al.* , 2001):

Area metrics: Area in the landscape; Landscape similarity metrics; Class area; Percent of Landscape; Total landscape area; Largest patch index.

Patch density, patch size and variability metrics: Number of patches; Patch density; Mean patch size; Patch size standard deviation; Patch size coefficient of variation.

Edge metrics: Perimeter; Edge contrast index; Total edge; Edge density; Contrast-weighted edge density; Total edge contrast index; Mean edge contrast index; Area-weighted mean edge contrast index.

Shape metrics: Shape index; Fractal dimension; Landscape shape index; Mean shape index; Area-weighted mean shape index; Double log fractal dimension; Mean patch fractal dimensions; Area-weighted mean patch fractal dimension.

Core area metrics: Core area; Number of core areas; Core area index; Core area percent of landscape; Total core area; Number of core areas; Core area density; Mean core area per patch; Patch core area standard deviation; Patch core area coefficient of variation; Mean area per disjunct core; Disjunct core area standard deviation; Disjunct core area coefficient of variation; Total core area index; Mean core area index.

Nearest-neighbour metrics: Nearest-neighbor distance; Proximity index; Mean nearest neighbour distance; Nearest-neighbor standard deviation; Nearest-neighbor coefficient of variation; Mean proximity index.

Landscape processes are generally inferred using species or community indicators, such as by modelling single species dispersal and persistence in the landscape, or correlating landscape patterns with community metrics of biodiversity such as species richness and turnover. Recent work has illustrated how some pattern metrics are highly correlated with biodiversity processes in terms of species and communities (Fischer & Lindenmayer, 2007). Particularly important metrics at the ecosystem and landscape scale are the amount and condition of natural vegetation, the number of anthropogenic edges (Harper *et al.* , 2005), and the configuration of patches (Fischer & Lindenmayer, 2007). However, there is still a need to further identify the abiotic and biotic processes that *cause* landscape patterns to emerge such as human land use, environmental change or natural disturbance, quantify the effect of patterns on functional and compositional biological processes, and elucidate the importance of such patterns and heterogeneity in maintaining ecosystem and landscape biodiversity (Turner, 2005; Fischer & Lindenmayer, 2007).

The above mentioned pattern metrics are generally based on discreet classifications of land use, and continuous variables may also be used to characterize effects on biodiversity. These include continuous vegetation cover, patterns in processes such as primary productivity, carbon or nitrogen mineralization and other environmental variables (Turner, 2005). Abiotic functional attributes of biodiversity are often estimated using remote sensing at the ecosystem level combined with environmental, geomorphic or hydromorphic modelling, or extrapolating from local data across ecosystem or biomes (e.g. Bouwman *et al.* , 2002; Elser *et al.* , 2007; Bobbink *et al.* , 2010).

The most frequent approach towards modelling biodiversity loss at ecosystem and landscape scales has been through the species area relationship (Arrhenius, 1921; Rosenzweig, 1995). Recent techniques are incorporating landscape pattern metrics into the SAR in order to represent the biodiversity value of non-natural habitat (Pereira & Daily, 2006), to account for fragmentation and patch size (Nelson *et al.* , 2009, 2008) and to reflect species' varying affinities to different landscape elements (Koh & Ghazoul, 2010a). This is by definition deductive, and relies on assumptions drawn from meta-study, landscape modelling or expert judgement. Results may be tested for accuracy using observed extinctions in empirical data (e.g. Koh & Ghazoul, 2010a; Kinzig & Harte, 2000).

8.1.6 Integrative indicators

Indicators that combine information across multiple attributes or components of biodiversity are termed integrative or multimetric indicators (EPA, 2003, 2008; Niemi & McDonald, 2004; Karr, 1981; Karr & Chu, 1997). Integrative indicators mathematically aggregate or weigh different attributes of biodiversity and express this in a single output. The Index of Biotic Integrity (IBI) was developed to monitor the health of aquatic ecosystems in the US using compositional and functional indicators including species richness, functional groups and indicator taxa sensitive to stress Karr (1981). It has since been taken up by the US EPA (2002) for wider national use, and adapted to numerous ecosystem types and taxa including birds and landscape diversity (O'Connell *et al.* , 2000) and coral reefs (Jameson *et al.* , 2001). Parkes *et al.* (2003) "habitat hectares" approach additively combines multiple indicators at the species and community level across compositional, functional and structural attributes. A similar aggregation and weighing approach is used in assessing WWF Ecoregion biological value and conservation status on an ecosystem level (Olson & Dinerstein, 1998). This produces a Biological Distinctiveness Index (BDI) reflecting biological value, and Conservation Status Index (CSI) reflecting pressures and vulnerability. Likewise, Important Bird Areas (IBAs) are monitored by aggregating biodiversity indicators across attributes (habitat area, quality and number of populations and trends of threatened "trigger" species) and weighing this against aggregated pressure indicators (agricultural intensification, pollution trends, population density and land use change; Bennun *et al.* , 2005). One potential setback of integrative indicators is that they rely on a subjective scoring system for different attributes and therefore must be tested extensively Karr & Chu (1997). However, the ability to not only reflect declines in biotic condition, but also diagnose potential causes makes them a very powerful as monitoring tools. They can also be aggregated and combined across multiple spatial scales (Niemi & McDonald, 2004).

8.1.7 The use of indicators in biodiversity assessments

We reviewed biodiversity assessment approaches in four models that have emerged recently to inform decision makers about the ecological consequences of public and private policy. We believe these models can serve as an example for how LCIA might develop in the future because they are 1) spatially explicit to some degree, 2) they attempt to model many pressures of biodiversity loss simultaneously, and 3) they demonstrate the use of the indicators and approaches discussed above. However, these models are incomplete with regards to modelling biodiversity, and in some cases even more limited than LCIA for certain pressures (such as ecotoxicity). Two of the models, the Millennium Ecosystem Assessment and the GLOBIO3 model, are global modelling tools, and therefore rough in the scale of impact assessment. The other two, the InVEST tool from the Natural Capital Project and the BioScore tool, are regional in their application. They therefore provide a selection of approaches towards dealing with issues of scale and data availability in biodiversity assessment. We believe this is particularly relevant to LCA which is positioned in a transition between site-generic and site specific impact factors (Hauschild & Potting, 2005) and between regional and global coverage (Bare, 2009).

The Millennium Ecosystem Assessment. The Millennium Ecosystem Assessment (2005a) modelled biodiversity loss on a global scale due to five direct drivers: terrestrial and aquatic habitat change, climate change, pollution, invasive species, and overexploitation. Terrestrial habitat change, i.e. land use, was modelled using a very coarse ecosystem indicator. Reductions in biome area were modelled according the four scenarios. Species richness of vascular plants, a community indicator, was extrapolated from local samples to estimate original biome species richness in each realm (Olson *et al.* , 2001). The species area relationship was then used to estimate past and future extinctions according to the scenarios (Van Vuuren *et al.* , 2006). Changes in biome coverage were calculated using the Integrated Model of Global Environmental Change (IMAGE; MNP, 2006) through four global policy scenarios (Van Vuuren *et al.* , 2006). It relied heavily on the species area relationship (SAR) and differentiated between local extirpation and regional/global extinctions. Local extirpation was considered reversible, but global losses were not (i.e. losses predicted by the SAR as habitat area declines to low levels). In a separate analysis, but using the scenarios of the MA, Jetz *et al.* (Jetz *et al.* , 2007) used range maps of all known bird species (Orme *et al.* , 2006) to model the effects of climate change and land conversion on global avian diversity. This used a species-level approach by calculating the extinction risk for each species due to range reductions.

The effects of water abstraction on biodiversity were modelled in the MA using catchment discharge rate as a predictor of species richness of fish across 237 river basins worldwide (Oberdorff *et al.* , 1995; Poff *et al.* , 2001; Xenopoulos *et al.* , 2005). This was employed to create a cause-effect relationship between water use which was estimated with the WaterGap model (Alcamo *et al.* , 2003) and species loss. The species-discharge relationship is convenient because it resembles the SAR in shape (power function where the marginal loss of species increases exponentially as water discharge declines to zero). River discharge approximates river basin ecological space and habitat heterogeneity, producing a species-discharge relationship that resembles the SAR (Xenopoulos *et al.* , 2005). Since global data on fish endemism is not available, only extinctions from individual river basins were considered, i.e. regional impacts in LCA terminology (MA, 2005a). The model was updated with recent data from FishBase (http://www.fishbase.org/). Alcamo *et al.*'s (2003) WaterGAP model was used to estimate future discharge rates due to water abstraction. A lack of data for other taxa (e.g. crustaceans, mussels, invertebrates), and a lack of a cause-effect models for aquatic habitats and vegetation communities (including Ramsar sites; www.ramsar.org) calls for more work to be done in this area (MA, 2005a).

The effects of climate change on terrestrial and aquatic biodiversity was incorporated via the above methods to estimate land and water use. Biome contractions and river discharge changes expected over the IPCC climate change scenarios was combined with the species area and species discharge relationship (for plants and fish respectively; (Van Vuuren *et al.* , 2006; Xenopoulos *et al.* , 2005)). The marine effects of climate change were limited to qualitative estimates. These include loss of corals and calcium dependent organisms from rising sea temperature and declining pH (Hughes *et al.* , 2003; Pandolfi *et al.* , 2003).

Acidification and eutrophication was covered in the MA (2005a) using meta-analyses of empirical studies to construct the cause-effect relationship between deposition of acidifying substances and species loss. Bouwman *et al.* (2002) constructed a global map of critical load values for both acidification and eutrophication based on FAO soil and GLC2000 land cover data. Exceedance ratios were calculated for each geographic region based on modelled deposition of sulphur and nitrogen on a 0.5° x 0.5° grid size (S72). The relationship between nitrogen exceedance and species richness of a variety of taxa across a ecosystem types and habitats was used to quantify species loss (Bobbink *et al.* , 2010; Schindler *et al.* , 1985; Bobbink *et al.* , 1998; Haddad *et al.* , 2000; Stevens *et al.* , 2004; Bobbink, 2004; Vinebrooke *et al.* , 2003). There is generally a non-linear relationship between CL exceedance and species richness (Bobbink, 2004; Alkemade *et al.* , 2009). Freshwater and marine ecosystems were omitted from quantitative analyses of acidification and eutrification in the MA as current knowledge is generally qualitative in nature (Elser *et al.* , 2007; Elser & Urabe, 1999; Downing & McCauley, 1992; Smith, 2003b). Further development of dose-response models based on meta-analysis of existing studies might yield appropriate damage factors (but see Weijters *et al.* , 2009).

Marine fisheries depletion was used as a proxy for overexploitation in the MA (S93). It employed regional case-studies and ecosystem models (the Ecopath and Ecosim fisheries models, http://www.ecopath.org/). Quantitative damages to marine biodiversity in three regions (Gulf of Thailand, coastal shelf; Benguella Current, upwelling; Central North Pacific, pelagic) were extrapolated across the globe (MA, 2005a). The estimations relate species losses via biomass declines in exploited trophic groups (functional indicator) to species losses (compositional indicator) via an index, Kempton's Q (Kempton, 2002). Other marine ecosystems have only been quantitatively described (deep sea, Glover & Smith, 2003; polar, Clarke & Harris, 2003; and vents/sea mounts, Koslow *et al.* , 2001). Finally, the impact of invasive species was assessed qualitatively across the MA scenarios, using globalized trade as a proxy for invasion risk. Expert opinion was used to gauge the importance of the driver in context to other drivers of biodiversity loss.

IMAGE 2.4 and the GLOBIO3 model. The GLOBIO3 model (Alkemade *et al.* , 2009) is an integrated, spatially explicit global assessment tool to investigate biodiversity change due to six terrestrial drivers of loss: land use, infrastructure, overexploitation, fragmentation, nitrogen deposition (pollution) and climate change. It is the latest tool to be implemented into the IMAGE model of environmental and economic change. It uses meta-analysis of published studies to establish cause-effect relationships under a species-level, intactness index called the Mean Species Abundance (MSA). The MSA is defined as the average fractional abundance decline of native species relative to their abundance in a pristine environment (Alkemade *et al.* , 2009). In actual fact, the MSA is approximated by species, community and ecosystem indicators according to the driver investigated. For land use, infrastructure and overexploitation, species-level change (intactness) is taken as a proxy for diversity. For fragmentation and nitrogen deposition, percentage loss of the community is used (community-level). For climate change, ecosystem-level effects on species richness are expressed via the SAR, combined with species-level measures using climate-envelope modelling and range size changes (Alkemade *et al.* , 2009).

The effects of land use and infrastructure was modelled using the MSA in each grid cell, and meta-study results to assign MSA values to each land use type. Fragmentation is further included by applying a weight on the MSA of land patches the fall below a critical minimum area of 10^3 km^2 (from Verboom *et al.* , 2007). Water use was not considered. Climate change compared two approaches: large scale biome area changes and species loss via the SAR in a similar approach to the MA (Leemans & Eickhout, 2004; Van Vuuren *et al.* , 2006), and species-based distribution modelling using EUROMOVE (Bakkenes *et al.* , 2002, 2006). MSA values are approximated by loss of species due to habitat loss or range size reduction. Pollution was modelled in the same was as the MA, using critical loading (CL) values for nitrogen deposition, and a meta-analysis of species loss due to CL exceedance (see above). Overexploitation of terrestrial species was considered a form of land use. Impact zones were established around infrastructure and roads in areas prone to overharvesting (e.g. tropical and sub-tropical forests) which effectively represent modifications to the existing land use classes. This reduces the biodiversity value (expressed as MSA) of the existing land class around such infrastructure. Invasive species were not considered. The GLOBIO3 model (Alkemade *et al.* , 2009) did not consider aquatic biodiversity loss, but will include both freshwater environments and marine aquatic ecosystems in the future, using a specific set of drivers of aquatic biodiversity loss (see http://www.globio.info/).

The Natural Capital Project's InVEST modelling tool. The InVEST tool (Nelson *et al.* , 2009, 2008) estimates changes in landscape biodiversity and ecosystems services resulting from only one driver of biodiversity loss: land use change. The model is applicable on an Ecoregion scale (Olson *et al.* , 2001) and utilizes as indicator species prominent macro-vertebrates or any higher order species group where regional data is available. Biodiversity loss is based on species- and ecosystem-level indicators of suitable habitat area using the HSI concept. Countryside SARs (Pereira & Daily, 2006) are used to aggregate habitat area on a per species basis and biodiversity is expressed as the ratio of existing habitat given a particular land use pattern to maximum potential area under natural conditions. Indefinite persistence in the landscape is estimated through the countryside SAR of each species (Nelson *et al.* , 2009). Additionally, dispersal ability and spatial configuration of habitat may be integrated into the model using landscape pattern metrics of configuration and fragmentation (see above).

The BioScore tool for European biodiversity assessment. The BioScore tool Louette *et al.* (2010) is a recently developed spatial biodiversity model designed to predict broad policy impacts on the state of European biodiversity. It assessed the effect of multiple pressures on species persistence across Europe. In total, 37 pressures are available in the BioScore database across categories including land use change, pollution, water quality and availability, climate change, fragmentation, disturbance, direct harvesting, interspecific interactions, and forest management (Delbaere *et al.* , 2009). It utilizes a species-level approach and incorporates a wide range of "focal species" (Lambeck, 1997) across mammals, reptiles, amphibians, birds, butterflies, vascular plants and freshwater fish (Louette *et al.* , 2010). The tool was designed around the "Driver-Pressure-State-Impact-Response (DPSIR) framework" (Spangenberg *et al.* , 2009) to translate the effects of policy decisions (Driver) in changing environmental variable (Pressure). This has an effect on the distribution of sensitive species (State) which causes an impact to biodiversity (Impact). Sensitivity scores and HIS are assigned to species using deductive methods (i.e. life history information and expert opinion) and four discreet sensitivity classes (using the criteria of Maes & Van Dyck, 2005). This information is used to model distributional changes across Europe due to broad scale policy decisions, and the concurrent effects on species persistence and extinction risk.

8.1.8 Representing the missing drivers of biodiversity loss

Two drivers of biodiversity loss in the MA, invasive species (biotic homogenization) and overexploitation (biotic depletion), are currently not represented in LCA.

Biotic depletion. Overexploitation of wild populations is recognized as one of the principal threats to global biodiversity across both aquatic and terrestrial ecosystems MA (2005a). In LCA, this is addressed as biotic depletion. A first conceptual approach to incorporate biotic depletion in LCA was based on the deaccumulation rate and total size of wild populations (Guinée *et al.* , 2002), and applied to case studies of fisheries (Ziegler *et al.* , 2003; Nilsson & Ziegler, 2007). A global approach covering a large range of species, and including indirect effects on the food chain is currently lacking.

The MA used regional case-studies and ecosystem models (Ecopath with Ecosim, http://www.ecopath.org/) to quantify species loss to marine biodiversity in three regions covering major oceanic zones, and then extrapolated across the global marine environment (MA, 2005a). Biodiversity loss was estimated via a functional indicator, the biomass declines in exploited trophic groups. Kempton's Q index (Kempton, 2002) was used to convert biomass declines into the compositional indicator of species loss. Other marine ecosystems were only qualitatively described. Halpern *et al.* (2008) constructed global, quantitative maps of marine overexploitation due to fishing that could guide impact factor construction.

For terrestrial ecosystems, the GLOBIO3 model (Alkemade *et al.* , 2009) and the Biodiversity Intactness Index (Scholes & Biggs, 2005) both consider harvesting as a form of land use. The former maps impact zones around infrastructure and roads in areas prone to overharvesting. The latter classifies natural areas into moderately used and degraded status. Both approaches use species level intactness indices, expressing the population size of species as a function of their abundance in pristine areas, and use regional data and remote sensing to define ecosystems and land use patterns. Such an approach could be employed in LCA to adjust regionalized land use methods to account for terrestrial biotic depletion.

Biotic homogenization. Invasive species influence the composition and species richness of exposed ecosystems, rivalling other global drivers of biodiversity loss (Clavero & García-Berthou, 2005). Cause-effect modelling of invasive species as a *separate impact category* is problematic in the current framework of LCA because invasion occurs through isolated outbreaks rather than continuous marginal impacts. Indirect incorporation through other impact categories is an option (Jolliet *et al.* , 2004). Land use impacts that take species turnover into account may weigh the presence of invasive species with zero, or negative, value (Jolliet *et al.* , 2004).

Targeting dispersal pathways as a proxy for invasion risk is a second approach. The MA (2005a) used global trade as a coarse proxy for invasion risk based on expert opinion. Recent introductions of aquatic invasive species have predominantly occurred through the release of biocontaminated ballast water, fouling of ship hulls, and the creation of waterways (Ricciardi & MacIsaac, 2000). Terrestrial transport networks and other land classes act as transport vectors for invasive plant species (Rodríguez-Labajos *et al.* , 2009). To establish cause-effect relationships, a search of relevant studies is needed that (i) link inventory flows with increased invasion risk, and (ii) express species loss as a function of invasion risk. The first requirement could be based on studies that correlate the prevalence of invasive species with inventory flows (i.e. construction of transport networks or transport distances). The second could use meta-analysis of empirical studies, such as Levine *et al.* (2003).

8.1.9 References

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8.2 Appendices for Chapter 3

8.2.1 Expanded methods

Definitions and literature search. The term “secondary growth” refers to vegetation communities in a process of recovery towards a climax stage following past disturbance. Our definition of disturbance was broad, and ranged from light impacts (e.g. selective logging, fire, grazing of natural grasslands) to total clearance of all vegetation (e.g. clear-cut logging, intensive livestock stocking, conversion to agriculture). Due to the variation in disturbance, we included previous disturbance intensity and duration as a predictor of diversity in our statistical models (see below). Old growth refers to climax vegetation with little or no signs of past disturbance with a “characteristic assemblage of species” from the same life zone as the SG area, and represents a feasible restoration target given the prevailing conditions in the landscape (SER, 2004). The term OG was not always synonymous with “primary” or “pristine” habitat in areas with a long history of human colonization and a scarcity of old growth vegetation, such as in Central Europe (Kardol *et al.* , 2005), where identifying original floral and faunal communities is not possible. In such cases, reference habitat referred to old, climax, (semi-)natural vegetation of high conservation value and with no signs of recent anthropogenic disturbance.

To locate relevant studies for the meta-analysis, we conducted an initial search on 27 August 2009 using Google Scholar, with the entries *’biodiversity OR diversity OR ”species diversity“ OR “species richness” AND restor* OR reclamation OR regenerat* OR regrowth OR recover* OR succession’*. The search generated more than 27’500 results, of which the most relevant 5000 articles were screened. On 11 July 2011, we conducted an additional ISI Web of Science search to find any recent literature using the entry ’*biodiversity OR "species diversity" OR "species composition") AND (restor* OR "secondary habitat" OR regrowth OR regenerat* OR reclamation OR succession)*’. This additional search resulted in 8’650 results, of which the most relevant 1000 were screened.

Similarity indices. The Sorenson index expresses the similarity between samples of species from two separate communities as:

$$SOR = \frac{2a}{2a + b + c} \tag{8.1}$$

Where a is the number of shared species, and b and c are the number of species unique to the individual samples, respectively. The index ranges from 0 (no shared species) to 1 (identical assemblages) and essentially measures of the proportion of shared species. The Morisita-Horn index integrates information on the (relative) abundance of species alongside presence-absence data, and is resistant to undersampling because its value is largely determined by frequency changes in the most abundance species (Jost, 2006; Beck *et al.* , 2013). Similarity is expressed as:

$$MH = \frac{2\sum_1^S x_i y_i}{\left(\frac{\sum_1^S {x_i}^2}{(\sum_1^S x_i)^2} + \frac{\sum_1^S {y_i}^2}{(\sum_1^S y_i)^2}\right)\sum_1^S x_i \sum_1^S y_i} \tag{8.2}$$

Where x_i is the value (abundance or frequency) of species i in the first sample and y_i is value (abundance or frequency) of species i in the second sample. All sums (Σ) are understood as $i = 1$ to $i =$ S (the pooled number of species across both sites).

Site independence. To maintain site independence when comparing sites of the same treatment (i.e. old growth), site-site comparisons were removed if distance was below a threshold for broad taxonomic groups. For plants, ground-dwelling arthropods, arboreal arthropods and invertebrates, 200 m, for amphibians, reptiles and non-volant small mammals, 400 m, for flying arthropods, birds, bats, and large mammals, 500 m. Since comparisons between SG and OG habitat attempt to hold site characteristics constant (and therefore place sites close together), these thresholds were not applied to SG–OG comparisons.

Index of sampling effort. For each study and habitat class, a total measure of sampling effort, expressed in area equivalents (m^2), was calculated for each site. Differences in sampling effort were calculated for each sample comparison (SG–OG or OG–OG). Sites were considered collections of nested spatial (pseudo-)replicates, each temporally replicated. In cases of plots, transects or points distributed across a landscape with no clear clustering into sites (O'Dea & Whittaker, 2007; Powers *et al.* , 2009), it was assumed each sampling point represented an independent site with a single spatial replicate. In general, a single day was the unit of temporal replication, with the seasonal replication disregarded (i.e. assumed to be considered in the individual studies). In general, between-study differences are unimportant because only differences within studies were considered in this analysis. Total sampling effort was calculated as:

*Effective sampling area (m^2) = Spatial replicate area (m^2) * Spatial replicates (#) * Temporal replicates (#) * Number of sites (#)*

The average effective sampling area per replicate (standardized effort) was estimated according to sampling method as follows, with sampling method classification following Dent and Wright (Dent & Wright, 2009).

Random plots or quadrats: Each plot (quadrats, transect, réleves etc.) was considered to be a single spatial replicate within a sampling site, with repeat sampling of the same plot at different dates considered temporal replication.

Visual observation (transect or point): Each walking transect or point count was considered a single spatial replicate within a sampling site, with repeat sampling of the same plot at different dates considered temporal replication. For point counts and walking transects, the amount of time spent per unit area (e.g. duration of point count, walking speed through transect) was not considered. Radius of point counts and detection limits for mammal transect walks, when not provided in the study, was assumed to be 20 m according to the breakpoint in detection success of Smith *et al.* (1997).

Systematic searching (transect or point): Each contiguous search area was considered a single spatial replicate. In a similar manner to visual observations, the time spent searching was assumed to be considered by the authors as suitable for the particular taxon and habitat.

Trapping: In general, each trapping grid or trapping transect was considered to represent a single spatial replicate (rather than individual traps). Usually, half the spacing distance between traps was considered to be the radius of trapping success (used for a buffer width when calculating total sampling area of the grid). When not provided, or when spacing significantly exceeded common values defined below, generic values were used according to trapping method. For fruit traps (fruit-feeding butterflies) the radius of effect was considered to be 50 m, the spacing distance of Barlow et al (2007c). For baited ground traps (dung beetles and ants), 25 m sampling radius based on Larsen and Forsyth (2005). For ant bait traps, 5 m sampling radius in time-limited bait trapping (usually 10 mins). For small mammals, a maximum of 10 m radius was used. For flight intercept traps (flying beetles), area calculations were excluded, but we recorded the interception surface area as an index of sampling effort. For light traps (moths), we used visual radii reported in studies (where provided) and otherwise set a 30 m range according to Beck and Linsenmair (2000) and J. Beck (pers. comm.). For pitfall traps, we assumed sampling radius was 5 m. For live traps (small mammals), we took an estimate of 10 m radius for each trap, which conforms to common values (e.g. Medellín and Equihua 1998).

Mist netting: Mist netting studies were assessed based on net area (in m^2) with net arrays (i.e. configurations of nets) used as the spatial replicate unit (not individual nets). Differences in opening time were disregarded.

Fogging: Fogging and collection effort was converted to equivalent sampling area through a ratio of collection area of funnel traps or collection sheets and expected leaf area per unit of crown area. In eucalypt stands (Fabião *et al.* , 1997) and ash stands (Vertessy *et al.* , 1995), the total leaf area of a tree of 50 cm dbh ranges 150-200 and 200-250 m^2, respectively. Assuming the tree crown has a diameter of 10 m (ca. 78 m2), this would imply a ratio of canopy area to leaf area of about 2:1 or 3:1. In larger broadleaved species such as those found in the tropics, this ratio is likely higher. Therefore we assumed a ratio of 5:1 for the leaf surface area in relation to area sampled for falling insects via funnels or collecting sheets (the 3 fogging studies included in the analysis were from tropical broadleaved species). This modification was conducted to better approximate the actual habitat area sampled in fogging studies.

Litter plots: Exact area of litter sampled was taken as sampling area, with each litter plot as a spatial replicate.

Response ratio. We used a log-transformed response ratio. The log-transformation linearises the response ratio, rebalancing the influence of deviations in the numerator and the denominator. It also partially corrects the skewed distribution of raw ratios (Hedges *et al.* , 1999). Effects involving zero values in either SG or OG samples are undefined and were therefore removed from the analysis. Although it is recommended to weigh response metrics by the inverse variance in each study (thereby placing more weight on studies reporting variability), this requires adequate sample replication for both treatments, normally greater than 5 samples (Hedges *et al.* , 1999). This information was not available from many of the large-scale field-studies included in the analysis, therefore we chose to use an unweighed standardized effect (Benayas *et al.* , 2009). In any case, because we attempted to standardize data at the site level, differences in sample size was not a large problem (99.2% of all comparisons were at the site level), and we included the average number of sites in the comparison as a predictor variable for all relevant analyses. We specifically accounted for variability and uncertainty separately through our resampling algorithms (see below).

Predictor data and subset models. The data were subset to facilitate construction of three main models using GLMs (see Table 3.1 in main text). Model 1, the “patch model”, included eleven predictors (age, distance, elevation, biome type, latitude, patch connectivity of SG habitat, patch size of SG and OG habitat, restoration status, difference in sampling effort, and taxon) and was primarily designed to quantify the relative contribution of habitat patch dynamics while accounting for other factors. As a result of the high data requirements, sample size was lowest for this model. Model 2, the “distance model” contained eight predictors (age, distance, elevation, biome type, latitude, restoration status, difference in sampling effort, and taxon), and was also restricted to site-level studies where the sample size for distance between sites was maximized (at the expense of patch variables). Model 2 primarily investigated the influence of “distance decay” on response metrics based on species similarity (Soininen *et al.* , 2007). Model 3, the “general model” (ten variables) was identical to model 2, but omitted the distance predictor and further included previous disturbance type, realm and number of sites. Model 3 maximized the sample size, which facilitated a more detailed taxonomic classification, and was useful in investigating the relative influence of general anthropogenic, ecological and biogeographic influences on the diversity of SG habitat relationship.

For all categorical predictors, a minimum sample size of 3 studies was assumed for each category or level. To achieve this, the degree of detail (i.e. number of levels) was adjusted for both biome type and taxon. All quantitative predictors were checked for collinearity using correlation matrices and corrgrams Friendly (2002). All three models described above were applied to response ratios of species similarity (Sorenson and Morisita-Horn). Only model 3 (the general model) was applied to the alpha diversity indices (relative species richness and Fisher's alpha) because they were not the main focus of the analysis.

GLMs and model averaging. The AIC is a measure of the information lost when moving from reality to a simplified model of defined predictors. It does this by ranking models relative to the AIC score of the best model in a subset, which is assumed to best approximate reality. Crucially, AIC penalizes complex models (i.e. those with many predictors), so a good model according to AIC is both well-fitting *and* simple. Predictors were recombined in the selection algorithm using a genetic recombination algorithm set at default values, rather than an exhaustive search, given the complexity of the resampling process (Calcagno & de Mazancourt, 2010).

Because AIC is a relative measure, it is also important to examine the structural goodness of fit of selected models (%DE) and to justify the inclusion of each predictor variable based on sound theory or previous research (Burnham & Anderson, 2002; Symonds & Moussalli, 2011). The predictors included in the models (described above and in the main text) all had been highlighted as potentially influential factors in previous SG studies. For example, the intensity and duration of the preceding disturbance affects SG-OG similarity (Dent & Wright, 2009; Mwampamba & Schwartz, 2011), the isolation status and size of SG habitat patches is thought to reduce colonization rates and lower the likelihood of population establishment, thereby affecting diversity indices (Dent & Wright, 2009) and the idiosyncratic response of different taxa and functional groups to disturbance is will documented in secondary growth studies (Barlow *et al.* , 2007b; Chazdon *et al.* , 2009a).

Variable	**M1**	**M2**	**M3**	**Description**
Age	x	x	x	Age of the secondary habitat, log10-transformed.
Distance	x	x		Geographic distance between sample sites in a comparison (km).
Disturbance			x	Disturbance intensity prior to recovery of the site or sampling area, with four levels (Dent & Wright, 2009). Extensive transformation (ET): Habitat anthropogenically modified without a subsequent occupation phase (i.e. recovery begins directly after disturbance). Ecosystem structure remains largely intact (e.g. selective logging, fire). Intensive transformation (IT): Habitat anthropogenically modified without a subsequent occupation phase (i.e. recovery begins directly after disturbance). Underlying ecosystem structure heavily modified (e.g. clear-cut forestry, habitat clearance). Extensive occupation (EO): Habitat anthropogenically modified and subsequently occupied for an undetermined time (i.e. disturbance is long-term). Ecosystem structure remains largely intact (e.g. agroforestry, forest-gardens, extensive grazing in grassland habitat) or occupation phase relatively short (e.g. slash and burn agriculture). Intensive occupation (IO): Habitat anthropogenically modified and subsequently occupied for an undetermined time (i.e. disturbance is long-term). Ecosystem structure heavily modified and long occupation phase (e.g. conventional farming, pasture in forested areas).
Elevation	x	x	x	Average elevation of the study area (masl).
Biome	x	x	x	Biome type of the study: coniferous forest, moist broadleaf forest, transitional dry forest/woodland and open shrub/grassland vegetation
Latitude	x	x	x	Absolute latitude, in degrees from the equator. Exact values were included at the site level where available, otherwise study average used.
Patch SG connectivity	x			SG connectivity status to old-growth habitat; 1 = connected, 0 = unconnected (Dent & Wright, 2009).
Patch SG/OG size	x			SG and OG patch size on logarithmic scale. 1 = < 10 ha, 2 = 10 - 100 ha, 3 = 100 - 1000 ha, 4 = > 1000 ha (Dent and Wright 2009).
Restoration method	x	x	x	Restoration status of the site, either active restoration (Restor_active) if some measures was taken to assist recovery (e.g. planting of native or exotic shade species, removal of exotics, periodic thinning), or passive restoration (Restor_passive) where cessation of disturbance was the only factor (i.e. natural succession).
Realm			x	Biogeographic realm of the study, after Olson *et al.* (2001) but omitting Oceania. Levels consisted of Australasia (AA), Afrotropic (AT), Indo-Malay (IM), Neotropic (NT), Nearctic (NA) and Palearctic (PA).
Sampling effort	x	x	x	Difference in sampling effort between SG and OG habitat calculated as $\log_{10}$(SG-effort/OG-effort). Sampling effort calculated as the number of sites per habitat in the comparison multiplied by the number of spatial and temporal replicates per site multiplied by the standardized area per replicate (i.e. #sites*#replicate*replicate size). See expanded methods above for details.
Number of sites			x	The average number of sites in the diversity comparison.
Taxon	x	x	x	Broad taxonomic group. For models 1 and 2 a simplified classification was used (plants, birds, mammals, herpetofauna and invertebrates). For model 3, a detailed classification was constructed: primarily non-woody species, woody species, birds, mammals, herpetofauna, insects and other invertebrates.

Table A.8.1 – Recovery model predictors. Predictor data extracted from studies to use in the GLMs. Three separate data subsets were analysed with GLMs to produce three models (M1, M2, and M3), differing in the predictors that were included (marked with “x”).

Reference	Taxa	Biome	Realm	Country	Latitude (dec °)	Longitude (dec °)	Elevation (masl)	RG sites	OG sites	Rep. data?
Abbott *et al.* (2003)	Cockroaches, grasshoppers, crickets and spiders	Transition	AA	Australia	-34.115	116.369	270	15	1	no
Aerts *et al.* (2008)	Birds	Broadleaf	AT	Ethiopia	13.617	39.350	1900	5	10	yes
Aidar *et al.* (2001)	Plants	Broadleaf	NT	Brazil	-24.529	-48.686	550	2	1	no
Andersen (1993)	Ants	Transition	AA	Australia	-12.670	132.830	30	6	2	yes
Andersen & Nelson (1999)	NVSM	Transition	NA	United States	34.279	-114.121	120	2	1	no
Andersen *et al.* (2001)	Grasshoppers	Open	AA	Australia	-12.853	132.587	20	10	10	no
Andrade & Rubio-Torgler (1994)	Birds	Broadleaf	NT	Colombia	-1.236	-69.922	100	3	4	no
Aravena *et al.* (2002)	Plants	Broadleaf	NT	Chile	-43.122	-74.030	75	4	1	no
Barlow *et al.* (2007b)	Birds	Broadleaf	NT	Brazil	-0.913	-52.561	70	5	5	no
Barlow *et al.* (2007c)	Butterflies	Broadleaf	NT	Brazil	-0.913	-52.561	70	5	5	no
Bihn *et al.* (2008)	Ants	Broadleaf	NT	Brazil	-25.305	-48.682	130	6	3	yes
Blake & Loiselle (2001)	Birds	Broadleaf	NT	Costa Rica	10.417	-84.017	100	2	1	no
Bobo *et al.* (2006a)	Butterflies	Broadleaf	AT	Cameroon	5.379	9.185	250	6	6	no
Bowen *et al.* (2009)	Birds	Transition	AA	Australia	-27.636	149.9	280	31	51	yes
Bowman *et al.* (1990)	Birds, butterflies and reptiles	Broadleaf	AA	Papua New Guinea	-5.000	145.733	380	13	3	no
Bragagnolo *et al.* (2007)	Harvestmen	Broadleaf	NT	Brazil	-23.739	-47.074	980	12	4	yes
Buckney & Morrison (1992)	Plants	Transition	AA	Australia	-32.41	152.47	80	3	4	yes
Castro-Luna *et al.* (2007)	Bats	Broadleaf	NT	Mexico	17.608	-92.463	190	6	2	no
Chambers *et al.* (1994)	Plants	Open	NA	United States	42.8	-111.367	1620	6	1	no
Chapman & Chapman (1997)	Plants	Broadleaf	AT	Uganda	0.574	30.394	1500	2	2	yes

Table A.8.2 – Literature database. Data summary of the 108 studies included in the meta-analysis, with locality and elevation values averaged across all study sites. “Rep. data” refers to whether the study reported OG replication data. Numbers in square brackets (“[1]”) indicate a second regional replicate study region within the same publication (considered independent data). (NVSM = non-volant small mammals). Realm abbreviations: Australasia (AA), Afrotropic (AT), Indo-Malay (IM), Neotropic (NT), Nearctic (NA), Palearctic (PA).

Reference	**Taxa**	**Biome**	**Realm**	**Country**	**Latitude (dec °)**	**Longitude (dec °)**	**Elevation (masl)**	**RG sites**	**OG sites**	**Rep. data?**
Chung *et al.* (2007)	Beetles	Broadleaf	IM	Malaysia	5.603	117.644	60	1	1	no
Costa *et al.* (2010)	Ants	Open	NT	Brazil	-18	-43.349	1250	7	1	no
de Souza *et al.* (2008)	Amphibians	Broadleaf	NT	Brazil	-22.462	-52.123	210	6	3	yes
Dranzoa (1998)	Birds	Broadleaf	AT	Uganda	0.436	30.535	1400	1	1	yes
Eggleton *et al.* (1997)	Termites	Broadleaf	IM	Malaysia	4.967	117.800	180	1	1	no
Eilu & Obua (2005)	Plants	Broadleaf	AT	Uganda	-1.030	29.686	1750	4	2	yes
Ernst & Roedel (2005)	Amphibians	Broadleaf	AT	Ivory Coast	5.833	-7.333	200	4	6	no
Estrada *et al.* (1994)	NVSM	Broadleaf	NT	Mexico	18.518	-95.169	1100	12	35	no
Fang & Peng (1997)	Plants	Transition	IM	China	21.464	110.905	40	7	1	no
Faria (2006)	Bats	Broadleaf	NT	Brazil	-15.283	-39.067	50	6	12	no
Farwig *et al.* (2008b)	Birds	Broadleaf	AT	Kenya	0.286	34.879	1500	3	3	no
Fermon *et al.* (2005)	Butterflies	Broadleaf	IM	Indonesia	-1.209	120.142	880	6	6	yes
Fimbel (1994)	Large mammals	Broadleaf	AT	Sierra Leone	7.549	-11.348	130	1	1	no
Floren *et al.* (2001)	Ants	Broadleaf	IM	Malaysia	6.046	116.703	500	3	1	no
Fukushima *et al.* (2008)	Plants	Broadleaf	IM	Thailand	18.553	98.515	930	21	7	yes
Gardner *et al.* (2007b)	Reptiles and amphibians	Broadleaf	NT	Brazil	-0.913	-52.561	70	5	5	no
Gardner *et al.* (2008)	Beetles	Broadleaf	NT	Brazil	-0.913	-52.561	70	5	5	no
Gillison *et al.* (2003)	Termites	Broadleaf	IM	Indonesia	-1.079	102.099	60	1	1	no
Glor *et al.* (2001)	Reptiles	Broadleaf	NT	Dominican Republic	19.06	-69.586	150	18	2	no
Gollan *et al.* (2011)	Beetles	Broadleaf	AA	Australia	-32.221	150.896	170	9	3	yes
Gove *et al.* (2005)	Ants	Transition	NT	Mexico	19.583	-96.383	30	6	3	yes
Grau *et al.* (1997)	Plants	Broadleaf	NT	Argentina	-26.8	-65.383	1000	4	2	yes
Grove (2002)	Saproxylic beetles	Broadleaf	AA	Australia	-16.132	145.415	60	3	3	yes
Hawes *et al.* (2009)	Moths	Broadleaf	NT	Brazil	-0.913	-52.561	70	5	5	no

Table A.8.2 cont.

Reference	Taxa	Biome	Realm	Country	Latitude (dec °)	Longitude (dec °)	Elevation (masl)	RG sites	OG sites	Rep. data?
Heinen (1992)	Reptiles and amphibians	Broadleaf	NT	Costa Rica	10.403	-84.021	100	2	1	no
Hingston & Grove (2010)	Birds	Broadleaf	AA	Australia	-43.164	146.698	260	9	2	yes
Hopp *et al.* (2010)	Beetles	Broadleaf	NT	Brazil	-25.372	-48.567	300	18	3	no
House *et al.* (2006)	Ants	Broadleaf	AA	Australia	-27.012	150.716	310	6	2	yes
Ishida *et al.* (2005)	Plants	Broadleaf	PA	Japan	34.417	129.333	350	2	2	no
Johns (1991)	Birds	Broadleaf	NT	Brazil	-3.533	-64.967	60	2	1	no
Kanowski *et al.* (2006) [1]	Reptiles	Broadleaf	AA	Australia	-27.154	152.651	200	2	1	no
Kanowski *et al.* (2006) [2]	Reptiles	Broadleaf	AA	Australia	-17.278	145.657	700	2	1	no
Kardol *et al.* (2005)	Macro-invertebrates	Open	PA	Netherlands	52.048	6.02	50	26	3	yes
Kennard (2002)	Plants	Transition	NT	Bolivia	-16.75	-61.75	500	12	1	no
Kindscher & Tieszen (1998)	Plants	Open	NA	United States	39.032	-95.14	260	3	2	yes
Klein (1989)	Beetles	Broadleaf	NT	Brazil	-2.417	-59.833	100	3	3	no
Kritzinger & Van Aarde (1998)	Birds	Transition	AT	South Africa	-28.626	32.074	50	5	1	no
Lambert (1992)	Birds	Broadleaf	IM	Malaysia	4.233	116.467	210	1	1	no
Law & Chidel (2001)	Bats	Broadleaf	AA	Australia	-37.055	149.873	100	1	1	no
Li *et al.* (2011)	Epiphytes	Broadleaf	PA	China	24.063	101.278	2300	3	1	no
Lieberman (1986)	Reptiles and amphibians	Broadleaf	NT	Costa Rica	10.403	-84.021	60	1	1	no
Liebsch *et al.* (2007)	Plants	Broadleaf	NT	Brazil	-25.435	-48.814	60		in 2008 study	
Liebsch *et al.* (2008) (and contained studies)	Plants	Broadleaf	NT	Brazil	-25.038	-48.861	350	13	5	yes
Luja *et al.* (2008)	Reptiles	Transition	NT	Mexico	18.444	-89.104	150	36	36	no
MacGregor-Fors *et al.* (2010)	Birds	Conifer	NT	Mexico	19.399	-102.245	2750	2	1	no
Majer (1992)	Ants	Broadleaf	NT	Brazil	-21.85	-46.567	1250	11	2	yes
Marín-Spiotta *et al.* (2007)	Plants	Broadleaf	NT	Puerto Rico	18.017	-66.083	640	15	3	no
Marsden (1998)	Birds	Broadleaf	IM	Indonesia	-3.283	129.567	300	1	1	no

Table A.8.2 cont.

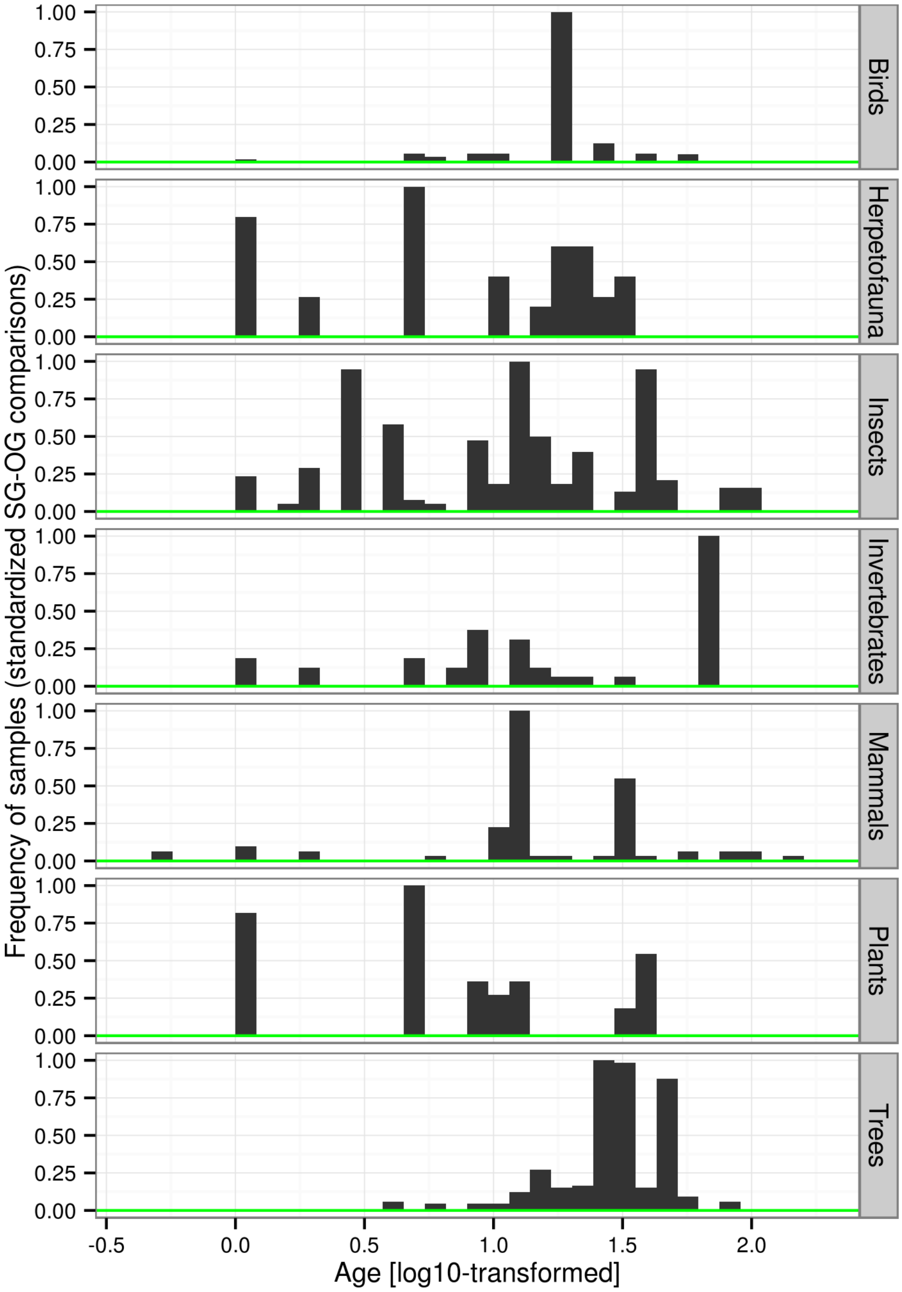

Figure A.8.1 – Age distribution of samples. Distribution of samples across time since disturbance (age) for each taxonomic group. Plot indicates frequency of comparisons across age (for count information see Appendix Table A.8.3). These comparisons were resampled for the GLMs.

Reference	**Taxa**	**Biome**	**Realm**	**Country**	**Latitude (dec °)**	**Longitude (dec °)**	**Elevation (masl)**	**RG sites**	**OG sites**	**Rep. data?**
McLauchlan *et al.* (2007)	Plants	Open	NA	Canada	49.867	-97.483	230	9	3	no
Medellín & Equihua (1998)	NVSM	Broadleaf	NT	Mexico	16.117	-90.933	120	4	4	no
Medellín *et al.* (2000)	Bats	Broadleaf	NT	Mexico	16.117	-90.933	120	3	3	no
Michael *et al.* (2011)	Reptiles	Transition	AA	Australia	-35.493	147.308	380	123	77	yes
Moola & Vasseur (2004)	Plants	Conifer	NA	Canada	44.083	-64.767	50	16	9	no
Nakagawa *et al.* (2006)	NVSM	Broadleaf	IM	Malaysia	4.198	114.043	175	9	5	yes
Nicolas *et al.* (2009)	NVSM	Broadleaf	AT	Guinea	8.33	-9.33	800	20	4	yes
O'Dea & Whittaker (2007)*	Birds	Broadleaf	NT	Ecuador	0.078	-78.684	1800	50	50	yes
Owiunji & Plumptre (1998)	Birds	Transition	AT	Uganda	1.75	31.5	1100	3	2	yes
Palladini *et al.* (2007)	Ants	Conifer	NA	United States	42.1	-123.2	1400	4	6	yes
Parrotta & Knowles (2001)	Plants	Broadleaf	NT	Brazil	-1.667	-56.45	180	5	1	no
Parry *et al.* (2007)	Large mammals and birds	Broadleaf	NT	Brazil	-0.913	-52.561	40	4	4	no
Pascarella *et al.* (2000)	Plants	Broadleaf	NT	Puerto Rico	18	-66.1	500	33	4	no
Patten (1997)	NVSM	Open	NA	United States	33.893	-116.573	230	1	1	no
Pawar *et al.* (2004)	Reptiles and amphibians	Broadleaf	IM	India	22.807	92.566	100	5	3	yes
Peh *et al.* (2005)	Birds (forest)	Broadleaf	IM	Malaysia	2.1	103.3	300	2	2	yes
Piqueray *et al.* (2011)	Plants	Open	PA	Belgium	50.104	4.91	250	12	8	no
Powers *et al.* (2009)*	Plants	Transition	NT	Costa Rica	10.84	85.62	550	22	3	yes
Purata (1986)	Plants	Broadleaf	NT	Mexico	18.582	-95.105	450	25	1	no
Quintero & Roslin (2005)	Beetles	Broadleaf	NT	Brazil	-2.417	-59.833	100	3	3	yes
Raman (2001)	Birds	Broadleaf	IM	India	23.566	92.388	500	10	2	yes
Renner *et al.* (2006)	Birds	Broadleaf	IM	Guatemala	15.383	-90.067	2250	6	6	no
Roth *et al.* (1994)	Ants	Broadleaf	NT	Costa Rica	10.403	-84.021	400	2	2	yes

* Study design deviated from well-defined sampling sites. For O'D&W 2007, single point counts distributed across a landscape, visited twice. For P 2009, similar design of plant plots. Each locality considered an individual site with low sampling intensity.

Table A.8.2 cont.

Reference	**Taxa**	**Biome**	**Realm**	**Country**	**Latitude (dec °)**	**Longitude (dec °)**	**Elevation (masl)**	**RG sites**	**OG sites**	**Rep. data?**
Sáfián *et al.* (2010)	Butterflies	Broadleaf	AT	Ghana	5.36	-1.369	180	4	3	yes
Sarmiento *et al.* (2003)	Plants	Broadleaf	NT	Venezuela	8.581	-70.7	900	115	8	no
Schonberg *et al.* (2004)	Ants	Broadleaf	NT	Costa Rica	10.333	-84.75	1450	1	1	no
Selmants & Knight (2003)	Plants	Conifer	NA	United States	41	-106	2900	30	24	no
Shahabuddin *et al.* (2005)	Beetles	Broadleaf	IM	Indonesia	-1.422	120.324	1150	4	4	no
Silva *et al.* (2007)	Ants	Broadleaf	NT	Brazil	-27.018	-49.017	575	5	1	no
Sodhi *et al.* (2005)	Birds	Broadleaf	IM	Indonesia	-1.107	120.128	1100	100	100	no
Sorensen & Fedigan (2000)	Large mammals	Broadleaf	NT	Costa Rica	10.833	-85.65	150	11	2	yes
Stark *et al.* (2006)	Plants	Conifer	NA	Canada	50.7	-120.3	900	21	10	yes
Stenbacka *et al.* (2010)	Beetles	Conifer	PA	Sweden	63.83	18.295	330	27	18	yes
Vallan (2002)	Amphibians	Broadleaf	AT	Madagascar	-18.954	48.522	900	2	4	yes
Vasconcelos (1999)	Ants	Broadleaf	NT	Brazil	-2.417	-59.833	80	3	2	yes
Veddeler *et al.* (2005)	Butterflies	Broadleaf	IM	Indonesia	-1.209	120.142	1100	24	4	no
Waltert *et al.* (2005a)	Butterflies	Broadleaf	IM	Indonesia	-1.209	120.142	1000	3	2	yes
Wijesinghe & Brooke (2005)	NVSM and birds	Broadleaf	IM	Sri Lanka	6.35	80.35	200	40	20	no
Willett (2001)	Spiders	Broadleaf	NA	United States	37.1	-122.13	520	2	1	no
Wu *et al.* (1996)	NVSM	Broadleaf	IM	China	22.025	100.871	700	6	7	yes

Table A.8.2 cont.

Taxon	Realm						Habitat		Biome		SG Restoration	
	AA	**AT**	**IM**	**NA**	**NT**	**PA**	**SG-OG**	**OG-OG**	**Forest**	**Open**	**Active**	**Passive**
a) Non-standardized comparisons												
All	3023	224	272	85	4184	166	5446	2508	7839	115	288	5158
Birds	2878	106	37	0	3560	0	4215	2366	6581	0	57	4158
Herpetofauna	18	15	36	0	37	0	88	18	106	0	5	83
Insects	92	13	21	39	192	7	320	44	349	15	82	238
Invertebrates (other)	15	0	0	2	54	81	143	9	71	81	0	143
Mammals	2	83	9	3	34	63	164	30	191	3	35	129
Plants (non-woody)	18	0	1	41	103	8	168	10	155	16	36	125
Trees	0	7	168	0	204	7	355	31	386	0	73	282
b) Standardized comparisons												
All	2967	207	242	70	3960	151	5085	2508	7496	101	250	4835
Birds	2875	99	31	0	3550	0	4189	2366	6555	0	50	4139
Herpetofauna	15	14	36	0	21	0	68	18	86	0	3	65
Insects	59	12	0	39	177	7	252	44	281	13	81	169
Invertebrates (other)	0	0	0	0	54	81	126	9	54	81	0	126
Mammals	0	82	7	0	26	63	144	30	178	0	32	112
Plants (non-woody)	18	0	0	31	0	0	39	10	42	7	24	15
Trees	0	0	168	0	132	0	269	31	300	0	66	203

Table A.8.3 – **Data summary**. Summary of the entire dataset including all comparisons (SG-OG and OG-OG) split into taxonomic groups for selected factors. Non-standardized comparisons (a) derived from studies where OG replicate data was not available, used for species richness and Fishers alpha models. Standardized comparisons (b) originated from studies which presented OG replicate data, used for Sorenson and Morisita-Horn models.

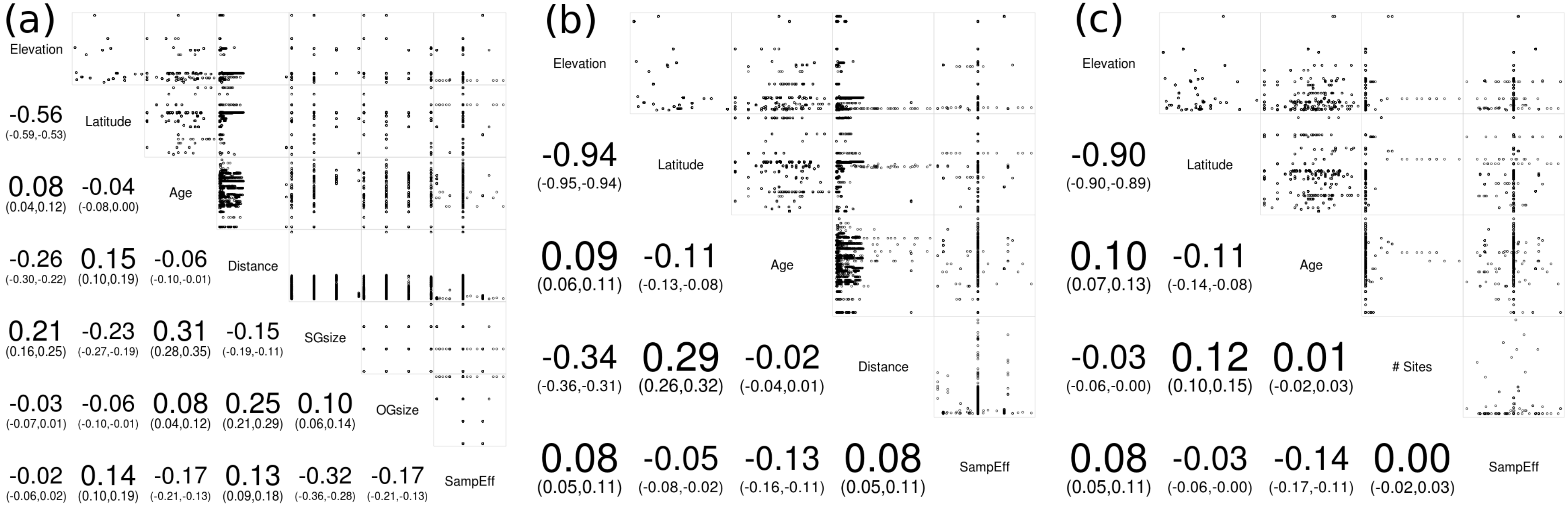

Figure A.8.2 – **Correlation matrices for predictor variables.** Correlation matrix for all quantitative parameters included in model 1-3 (plots a, b and c, respectively). Confidence intervals in parentheses. SGsize/OGsize = SG/OG patch size, SampEff = sampling effort difference.

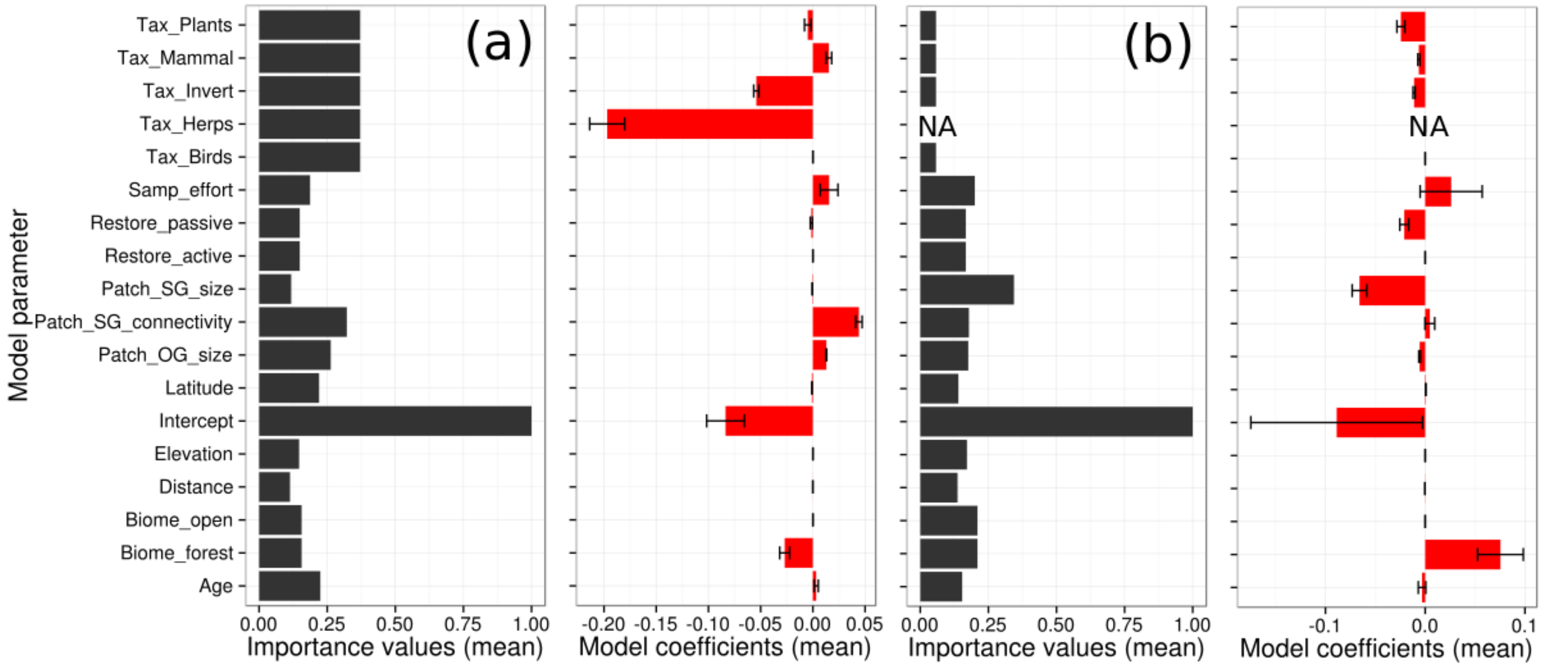

Figure A.8.3 – Model-averaged parameters of the patch model (model 1) illustrating the effects of patch dynamics on Sorenson (a) and Morisita-Horn (b) similarity. Model-averaged importance values (left) and coefficients +/- variance (right) for model 1 of Sorenson (a) and Morisita-Horn (b) similarity, illustrating effects of patch dynamics (size and isolation). Data on herpetofauna for M-H not available. Derived from N = 10'000 resampled model runs with replacement and sample sizes (# studies) of n = 18 (Sor) and n = 16 (M-H). NA = Not Available

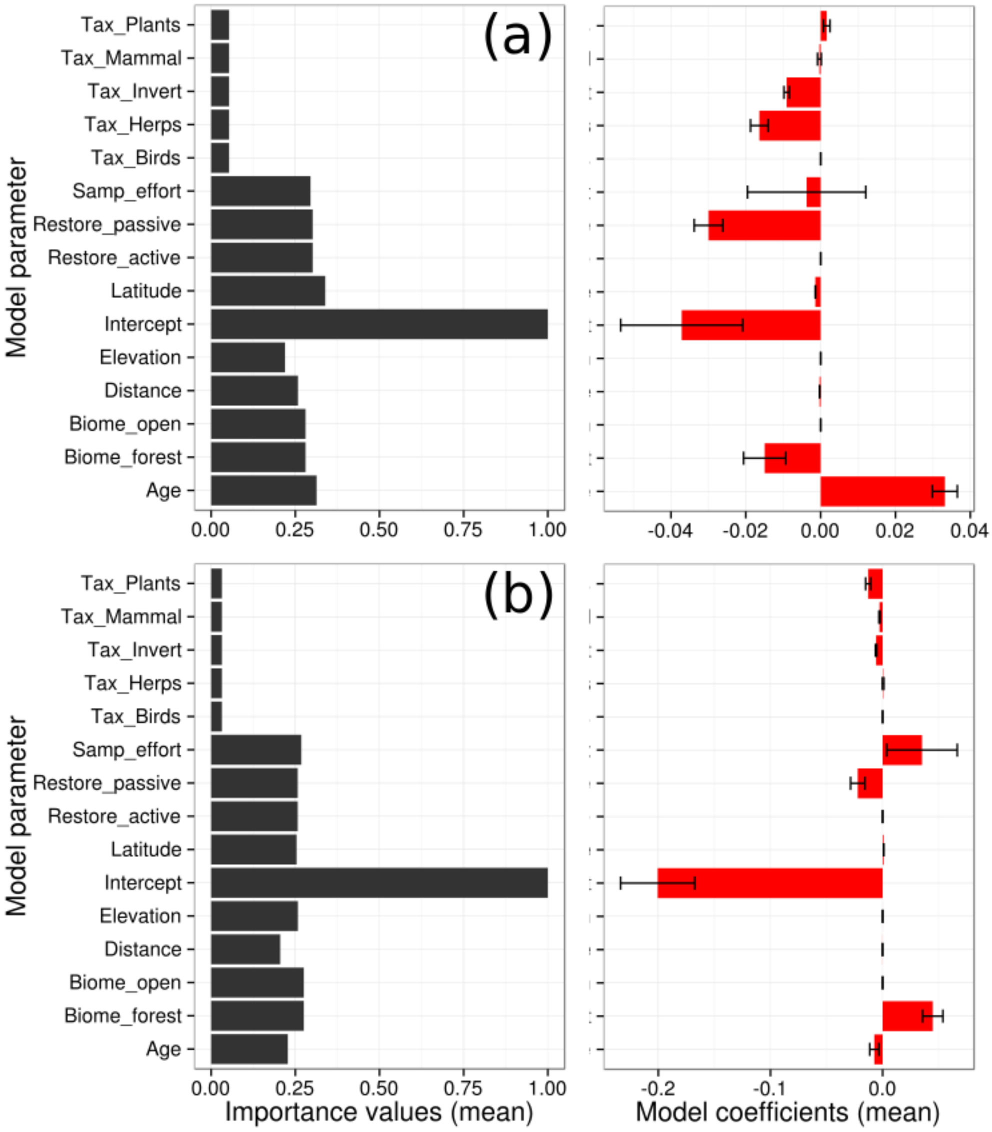

Figure A.8.4 – Model-averaged parameters of the distance model (model 2) illustrating the distance decay in Sorenson (a) and Morisita-Horn (b) similarity. Model-averaged importance values (left) and coefficients +/- variance (right) for model 2 of Sorenson (a) and Morisita-Horn (b) similarity, illustrating effects of including distance between samples as a predictor variable. Derived from N = 10'000 resampled model runs with replacement and sample sizes (# studies) of n = 25 (Sor) and n = 21 (M-H).

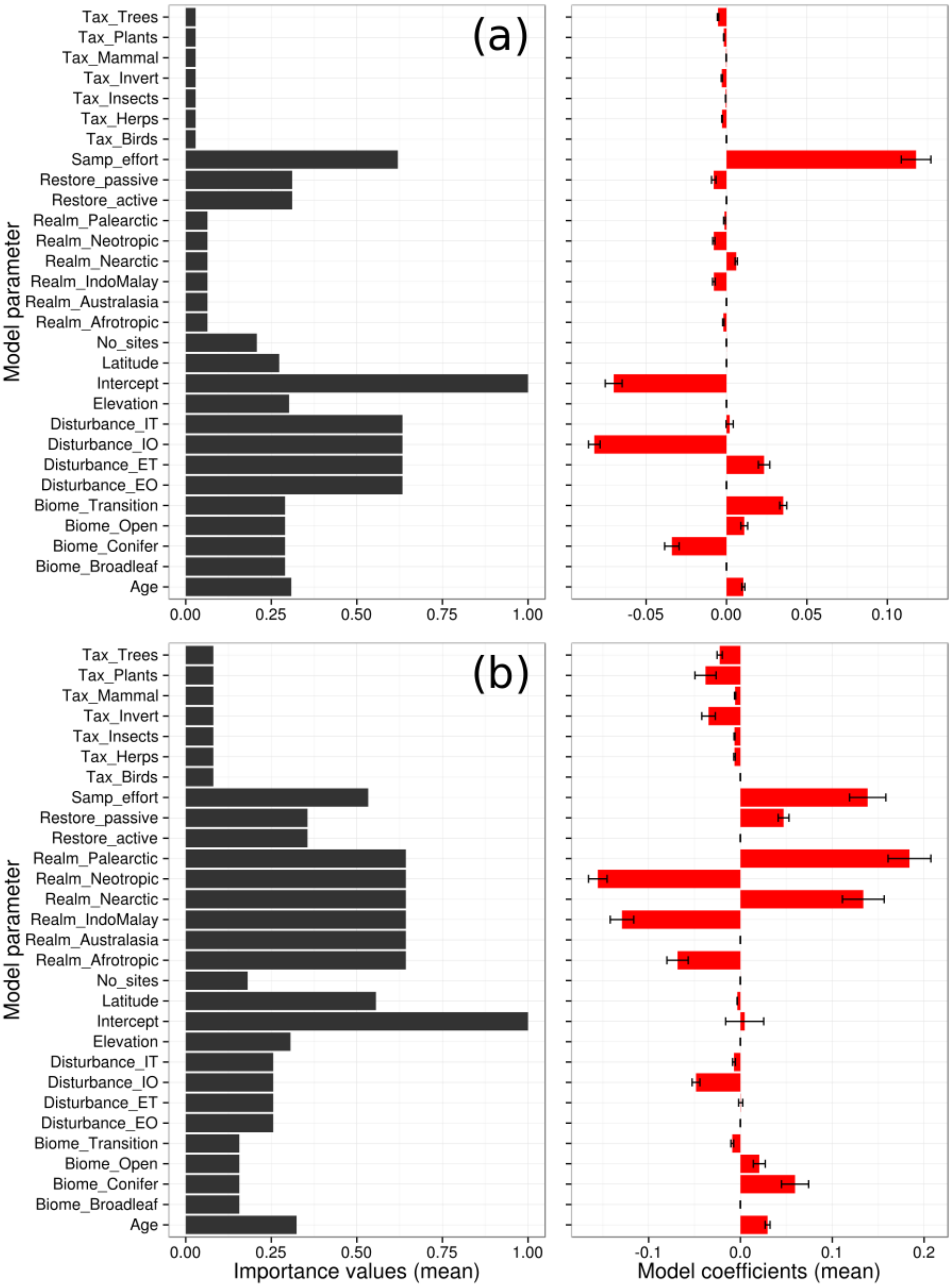

Figure A.8.5 – Model-averaged parameters of the general model (model 3) for species richness (a) and Fisher's alpha (b). Model-averaged importance values (left) and coefficients +/- variance (right) for model 3 of species richness (a) and Fisher's alpha (b), illustrating effects of general ecological and biogeographical factors. Derived from N = 10'000 resampled model runs with replacement and sample sizes (# studies) of n = 106 (S. R.) and n = 72 (Fisher).

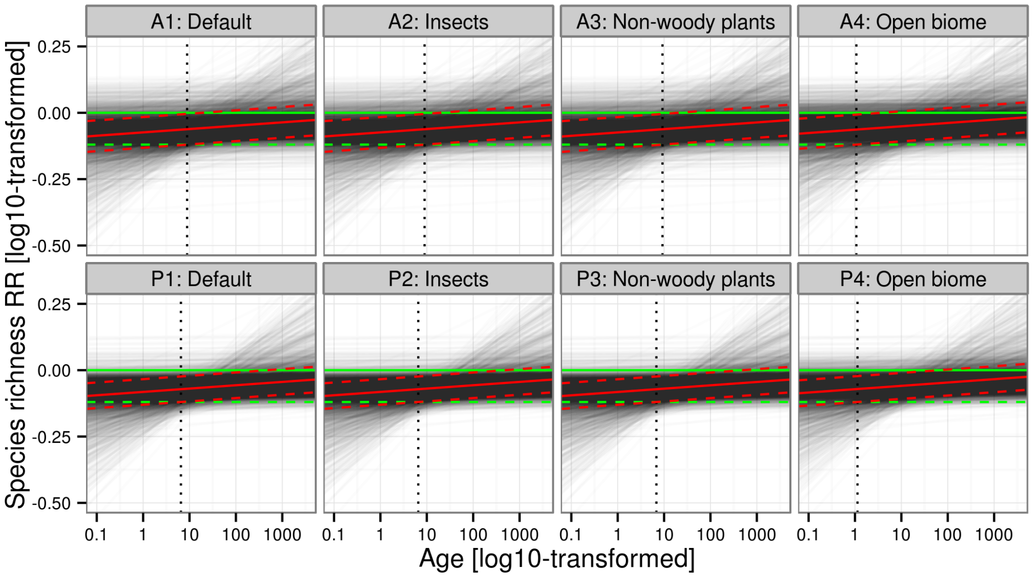

Figure A.8.6 – Modelled recovery trajectories for species richness. Partial effects of selected predictors on the recovery of relative species richness (RSR) for active (A1-4, upper row) and passive (P1-4, lower row). Individual regression lines for the resample runs are plotted with density shading = black when > 2.5% of data overlap. Mean model values and standard deviation as solid and broken red lines, respectively. Mean and standard deviation of OG–OG control values as solid and dotted green line, respectively. Facets represent changes to individual parameters of the default global model, which assumes taxon = birds, biome = broadleaf forest, realm = Australasia, disturbance = extensive occupation, and median dataset values for continuous variables.

Taxon	**B**	**R**	**Predicted average recovery time [y]**										
			AA	**AA**	**AT**	**AT**	**IM**	**NT**	**NT**	**NA**	**NA**	**PA**	**PA**
			(11)	**(22)**	**(11)**	**(22)**	**(11)**	**(11)**	**(22)**	**(33)**	**(45)**	**(33)**	**(45)**
Birds	F	P	59	55	74	68	93	92	85	50	47	68	64
Herps	F	P	67	63	83	77	106	104	97	57	53	78	72
Insects	F	P	60	56	74	69	94	92	86	50	47	69	64
Invert.	F	P	69	64	85	79	108	106	99	58	54	79	74
Mammals	F	P	60	56	74	69	94	93	86	51	47	69	64
Plants	F	P	64	59	79	73	100	98	91	54	50	73	68
Trees	F	P	88	82	109	102	139	136	127	75	69	102	95
Birds	F	A	12	11	15	14	19	19	17	10	10	14	13
Herps	F	A	14	13	17	16	22	21	20	12	11	16	15
Insects	F	A	12	11	15	14	19	19	18	10	10	14	13
Invert.	F	A	14	13	17	16	22	22	20	12	11	16	15
Mammals	F	A	12	11	15	14	19	19	18	10	10	14	13
Plants	F	A	13	12	16	15	21	20	19	11	10	15	14
Trees	F	A	18	17	22	21	28	28	26	15	14	21	19
Birds	NF	P	6	6	7	7	9	9	9	5	5	7	6
Herps	NF	P	7	6	8	8	11	11	10	6	5	8	7
Insects	NF	P	6	6	7	7	10	9	9	5	5	7	6
Invert.	NF	P	7	6	9	8	11	11	10	6	5	8	7
Mammals	NF	P	6	6	8	7	10	9	9	5	5	7	7
Plants	NF	P	6	6	8	7	10	10	9	5	5	7	7
Trees	NF	P	9	8	11	10	14	14	13	8	7	10	10
Birds	NF	A	1	1	2	1	2	2	2	1	1	1	1
Herps	NF	A	1	1	2	2	2	2	2	1	1	2	1
Insects	NF	A	1	1	2	1	2	2	2	1	1	1	1
Invert.	NF	A	1	1	2	2	2	2	2	1	1	2	2
Mammals	NF	A	1	1	2	1	2	2	2	1	1	1	1
Plants	NF	A	1	1	2	2	2	2	2	1	1	2	1
Trees	NF	A	2	2	2	2	3	3	3	2	1	2	2

Table A.8.4 – Species richness recovery times. Sensitivity analysis of recovery times for species richness for various model scenarios. Recovery times represent time required for SG-OG diversity to fall within $\frac{1}{2}$ a standard deviation of OG-OG reference values. Numbers in parentheses below realm indicate latitude (°). B = Biome Forest, F, Non-forest, NF), R = Restoration (Passive, P, Active, A.

Taxon	**B**	**R**	**Predicted average recovery time [y]**										
			AA	**AA**	**AT**	**AT**	**IM**	**NT**	**NT**	**NA**	**NA**	**PA**	**PA**
			(11)	**(22)**	**(11)**	**(22)**	**(11)**	**(11)**	**(22)**	**(33)**	**(45)**	**(33)**	**(45)**
Birds	F	P	122	193	124	195	125	128	202	127	200	129	203
Herps	F	P	124	195	125	197	126	130	204	129	202	131	206
Insects	F	P	137	216	139	219	140	144	227	143	224	145	228
Invert.	F	P	136	214	138	216	138	143	224	141	222	143	226
Mammals	F	P	127	200	129	203	129	133	210	132	208	134	211
Plants	F	P	105	165	106	167	107	110	173	109	172	111	175
Trees	F	P	138	218	140	221	141	145	229	144	226	146	230
Birds	F	A	49	78	50	79	50	52	82	51	81	52	82
Herps	F	A	50	79	51	80	51	53	83	52	82	53	83
Insects	F	A	56	87	56	88	57	58	92	58	91	59	92
Invert.	F	A	55	86	56	88	56	58	91	57	90	58	91
Mammals	F	A	51	81	52	82	52	54	85	53	84	54	85
Plants	F	A	42	67	43	68	43	45	70	44	69	45	71
Trees	F	A	56	88	57	89	57	59	92	58	91	59	93
Birds	NF	P	83	131	84	133	85	87	137	86	136	88	138
Herps	NF	P	84	132	85	134	86	88	139	87	137	89	140
Insects	NF	P	93	147	95	149	95	98	154	97	153	99	155
Invert.	NF	P	92	145	93	147	94	97	152	96	151	98	153
Mammals	NF	P	86	136	88	138	88	91	143	90	141	91	144
Plants	NF	P	71	112	72	114	73	75	118	74	117	75	119
Trees	NF	P	94	148	95	150	96	99	155	98	154	99	156
Birds	NF	A	34	53	34	54	34	35	56	35	55	36	56
Herps	NF	A	34	54	34	54	35	36	56	35	56	36	57
Insects	NF	A	38	59	38	60	38	40	62	39	62	40	63
Invert.	NF	A	37	59	38	59	38	39	62	39	61	39	62
Mammals	NF	A	35	55	35	56	36	37	58	36	57	37	58
Plants	NF	A	29	45	29	46	29	30	48	30	47	30	48
Trees	NF	A	38	60	39	61	39	40	63	39	62	40	63

Table A.8.5 – Sorensen similarity recovery times. Sensitivity analysis of recovery times for Sorenson similarity for various model scenarios. Recovery times represent time required for SG-OG diversity to fall within $\frac{1}{2}$ a standard deviation of OG-OG reference values. Numbers in parentheses below realm indicate latitude (°). B = Biome (Forest, F, Non-forest, NF), R = Restoration (Passive, P, Active, A).

Taxon	**B**	**R**	**Predicted average recovery time [y]**										
			AA (11)	**AA (22)**	**AT (11)**	**AT (22)**	**IM (11)**	**NT (11)**	**NT (22)**	**NA (33)**	**NA (45)**	**PA (33)**	**PA (45)**
Birds	F	P	358	382	339	481	375	400	497	530	374	400	173
Herps	F	P	2706	2888	2561	3630	2833	3023	3752	4004	2828	3018	1305
Insects	F	P	4455	4754	4217	5976	4664	4977	6177	6592	4656	4968	2148
Invert.	F	P	491	524	465	659	514	549	681	727	514	548	237
Mammals	F	P	11506	12279	10892	15435	12045	12854	15954	17025	12025	12832	5548
Plants	F	P	231	247	219	310	242	258	320	342	241	258	111
Trees	F	P	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Birds	F	A	337	299	319	424	331	353	438	468	330	353	152
Herps	F	A	2549	2261	2413	3204	2500	2668	3312	3534	2496	2664	1152
Insects	F	A	4196	3722	3972	5275	4116	4393	5452	5818	4109	4385	1896
Invert.	F	A	463	411	438	582	454	485	601	642	453	484	209
Mammals	F	A	10838	9614	10259	13623	10632	11345	14082	15027	10613	11326	4897
Plants	F	A	218	193	206	274	214	228	283	302	213	227	98
Trees	F	A	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Birds	NF	P	184	164	174	232	181	193	240	256	181	193	316
Herps	NF	P	1392	1235	1318	1750	1366	1458	1809	1931	1364	1455	2388
Insects	NF	P	2292	2033	2170	2881	2249	2400	2978	3178	2245	2396	3932
Invert.	NF	P	253	224	239	318	248	265	329	351	248	264	434
Mammals	NF	P	5921	5252	5604	7442	5808	6198	7693	8209	5798	6187	10156
Plants	NF	P	119	105	113	149	117	124	154	165	116	124	204
Trees	NF	P	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Birds	NF	A	163	144	144	154	205	160	170	211	226	159	170
Herps	NF	A	1229	1090	1090	1163	1545	1206	1287	1597	1704	1204	1284
Insects	NF	A	2023	1795	1795	1915	2543	1985	2118	2629	2805	1981	2114
Invert.	NF	A	223	198	198	211	281	219	234	290	309	219	233
Mammals	NF	A	5226	4635	4635	4947	6569	5126	5471	6790	7246	5118	5461
Plants	NF	A	105	93	93	99	132	103	110	136	146	103	110
Trees	NF	A	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Table A.8.6 – Morisita-Horn similarity recovery times. Sensitivity analysis of recovery times for Morisita-Horn similarity for various model scenarios. Recovery times represent time required for SG-OG diversity to fall within $\frac{1}{2}$ a standard deviation of OG-OG reference values. Numbers in parentheses below realm indicate latitude (°). B = Biome (Forest, F, Non-forest, NF), R = Restoration (Passive, P, Active, A). NA = Not Available

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8.3 Appendices for Chapter 4

8.3.1 Vegetation classification

We created 8 simple vegetation classes based on MODIS Vegetaion Continuous Field 250 m resolution percentage tree cover for 2010 (Hansen *et al.* , 2003) and elevation bands (SRTM 90 m resolution dataset). We adjusted the borders of classes based on personal knowledge of the study region and general vegetation patterns from the literature (Kindt *et al.* , 2007b). This lead to the following classes (see Figure A.8.1):

1. Low-elevation grassland and bush (% tree cov. < 20; elev < 2'200 m)
2. High-elevation grassland and bush (% tree cov. < 20; elev > 2'200 m & < 3'000 m)
3. Low-elevation sparse or degraded forest (% tree cov. > 20 & < 45; elev < 1'800 m)
4. Mid-elevation sparse or degraded forest (% tree cov. > 20 & < 45; elev > 1'800 m & < 3'000 m)
5. High-elevation woodland (% tree cov. > 20 & < 45; elev > 3'000 m)
6. Low-elevation intact closed-canopy forest (% tree cov. > 45 & < 60; elev < 2'400 m)
7. High-elevation intact closed-canopy forest (% tree cov. > 45 & < 60; elev > 2'400 m)
8. Alpine vegetation (% tree cov. < 20; elev > 3'000 m).

8.3.2 Household surveys

We used a standardized questionnaire to collect information relating to general demographic characteristics, community integration, land management, local land prices, property value, input labour and material costs for major crop and livestock types, product sales, yield and selling price, household expenditures, investments, incomes from off-farm employment and monthly profits from business activities (this information is presented in detail in Curran *et al.* , 2013, and an example questionnaire is presented at the end of the Appendix, Figure A.8.5). Of the 300 households, sampling was split into two regional blocks differing in agricultural potential, and stratified according to population density, with the first block consisting of 200 households in the densely populated districts of Muranga and Thikka, south-east of the Aberdares Conservation Area. A second block of 100 households was located in the districts of Nyeri and Nyandarau, north-east of the Aberdares Conservation Area (Fig. A.8.1). The sampling allocation roughly matches differences in population density between the regions which differ by a factor of 2 (according to the 2009 national population census, Nyandarau and Muranga had population densities of 191 pers/km^2 and 405 pers/km^2, respectively; www.wikipedia.org). Households were chosen along several transects along minor to

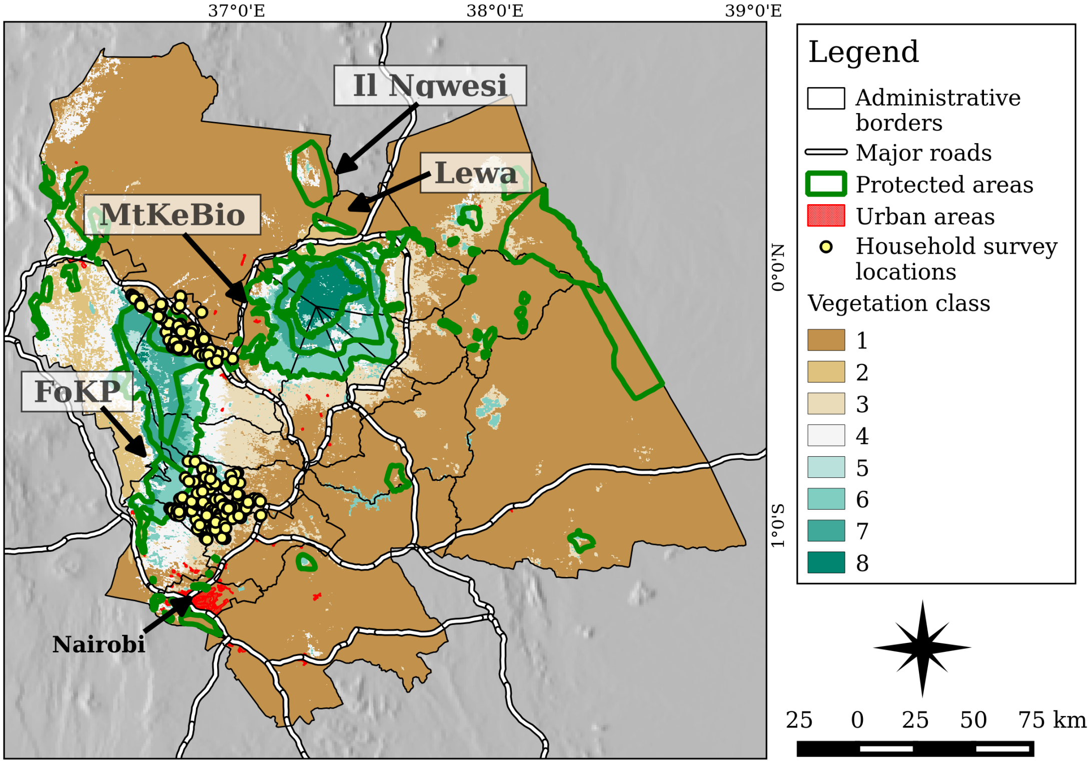


Figure A.8.1 – Map of the study area. Map shows locations of household surveys (points) and conservation organizations interviewed for management cost data (labelled arrows). Data overlaid onto a map of vegetation classes derived from elevation–tree cover combination (see Section 8.3.1). Beyond the regional extent, topography is represented by a hillshade elevation map (SRTM 90m resolution).

major roads, running from the Aberdare Forest Reserve boundary to lower elevations. Two teams travelled door to door on opposite sides of the road, and approached farms located at variable distance from the road (up to ca. 500 m). After locating a willing respondent and conducting one interview, both teams would move at least 200 m further along the road and repeat. Interviews were conducted by members of the Centre for Training and Integrated Research in Arid and Semi-Arid Lands Development (CETRAD), Nanyuki.

Survey results: per-household and per-hectare net returns

Most farms exhibited low specialization and diverse mixes of crops, pasture, fodder and zero or semi-zero grazing livestock systems that were difficult to analyse in isolation. Therefore, when estimating per-hectare returns (approximating opportunity costs), we bundled all production systems together and divided total agricultural net returns by the amount of agricultural land (crops, fodder and pasture). Returns from forestry or other production systems were ignored. We also asked respondents if they grazed livestock on

Variable	**Mean**	**Median**	**Min**	**Max**	**S.D.**	**%HH**
Number of family members (resident and non-resident)	10.1	7.0	1.0	57.0	8.5	99.7
Agricultural wage [$/month]	56.0	60.2	30.1	97.9	10.9	99.7
Livestock herder wage [$/month]	48.8	45.2	15.1	90.4	15.2	90.0
Distance to local market [km]	3.18	2.00	0.10	39.00	4.08	90.7
Distance to next larger market [km]	14.48	10.00	0.50	75.00	13.17	86.7
Land price [$/ha]	18’920.9	14’880.0	164.0	119’036.0	14’621.5	77.3
Property value [$]	19’396.7	10’241.0	133.0	140’964.0	22’678.8	77.3
Total land area [ha]	1.23	0.81	0.03	15.79	1.56	100.0
Area under crop cultivation [ha]	0.53	0.36	0.03	4.45	0.54	97.7
Area under pasture management [ha]	0.52	0.30	0.01	3.64	0.63	26.0
Area under fodder cultivation [ha]	0.26	0.10	0.0016	4.05	0.48	47.7
Area under exotic forestry [ha]	0.19	0.10	0.0016	0.81	0.19	24.0
Area under native forestry [ha]	0.21	0.10	0.03	1.21	0.25	7.3
Value of livestock herd [$]	1’275.3	848.8	4.2	11’515.7	1’375.7	96.0
Annual returns from off-farm employment by household residents [$/year]	2’922.5	2’168.7	180.7	10’120.5	2’383.5	45.0
Annual returns from off-farm employment by all family members (residents and non-residents) [$/year]	6’481.9	3’180.7	180.7	42’650.6	7’651.8	57.0
Net annual returns from business and petty trade [$/year]	1’508.0	867.5	72.3	7’228.9	1’578.0	13.3
Net annual returns from crop cultivation [$/year]	2’448.6	979.5	-1’372.9	93’164.3	6’247.1	97.0
Net annual returns from livestock [$/year]	-1’138.4	-698.3	-12’638.6	5’451.8	1’838.4	96.7
Net annual returns from livestock products (milk, eggs, hides etc.) [$/year]	1’210.5	838.6	-954.2	24’085.9	2’165.2	70.3
Toal net annual returns (crops, livestock and products combined) [$/year]	2’399.5	979.5	-11’199.6	93’144.9	6’474.3	100.0
Net agricultural opportunity costs [$/ha/y]	5’910.5	2’148.9	-55’325.9	121’448.2	13’545.1	100.0

Table A.8.1 – Household survey summary statistics. Note that crop and overall agricultural returns include weighing for the number of growing seasons. Summary statistics are calculated using different sample sizes due to variable response rates (%HH = percentage of households providing responses), therefore totals may not add up (i.e. crop + livestock + product != total net returns). S.D. = standard deviation.

common property or forested areas (which was observed during surveys) to exclude these effects on returns, but almost all households responded negatively. Therefore our net return estimates from livestock production might be biased upwards (i.e. larger stocks than possible on farm) or downwards (respondents omitted livestock that were not on farm at the time of the interview). Stated land prices were used directly from the survey results, and reflected landowner perceptions of current land prices in their locality.

Summary statistics for the surveys are provided in Table A.8.1. Interviews lasted on average 1 h 22 mins (S.D. = 21 mins). Land holdings in the study region were small, with a mean property size of 1.4 ha (median = 0.81, S.D. = 1.56) and a maximum of 15.79 ha (Appendices, Table A.8.1). 98.6% of respondents stated that they owned valid proof of land tenure (letter of allotment, title-deed or title-deed in processing). Nearly all landowners (97.7%) devote part of their land to crop cultivation, averaging 0.53 ha (median = 0.36, S.D. = 0.54). Pasturing of livestock and fodder production were less common (26% and 47.7% of households, respectively), but covered comparable land area (0.52 and 0.26 ha, respectively). A minority of households engaged in forestry, with exotic species most common (24% of households and an average area of 0.19 ha), but 7.3% of respondents indicated comparable acreage of native species (mean = 0.21 ha).

Income sources were divided across farming activities and off-farm employment, with 45% of households reporting returns from off-farm employment by farm residents (mean returns = \$2'922.5, median = \$2'168.7, S.D. = \$2'383). This figure rose to 57% of households when non-residents are considered (e.g. family members living away from home). Only 13.3% of households engaged in business activities, with a mean contribution to household returns of \$1'508 but a high variability (median = 876.5, S.D. = 1'578). Returns from non-farming activities generally exceeded that of farming. Net annual returns from crop cultivation at the household level averaged \$2,448.6 (median = 979.5, S.D. = 6'247.1), and almost all reported values were positive. Only 6% of respondents reported financial losses from crop cultivation, with most frequently cited reasons of abandonment and crop failure. Livestock returns were lower, despite the fact that a similar proportion of households engaged in some form of animal husbandry (usually for home consumption of milk, meat and eggs, rather than sale). Net returns from both the sale and consumption of products were negative in about two-thirds of surveyed households (61%).

All considered, combined net agricultural returns across crops, livestock and livestock products were \$2'399.5/y (median = 979.5, S.D. = 6'474.3), with 20% of households reporting negative agricultural returns. Corrected for area of cultivated land, per-hectare net returns (opportunity costs) averaged \$5'910.5 (median = 2'148.9, S.D. = 13'545.1). We detected a weak, but significant, *negative* correlation between log-transformed farm size and both ranked opportunity costs (n = 227, p = <0.001, R^2= 0.092) and ranked

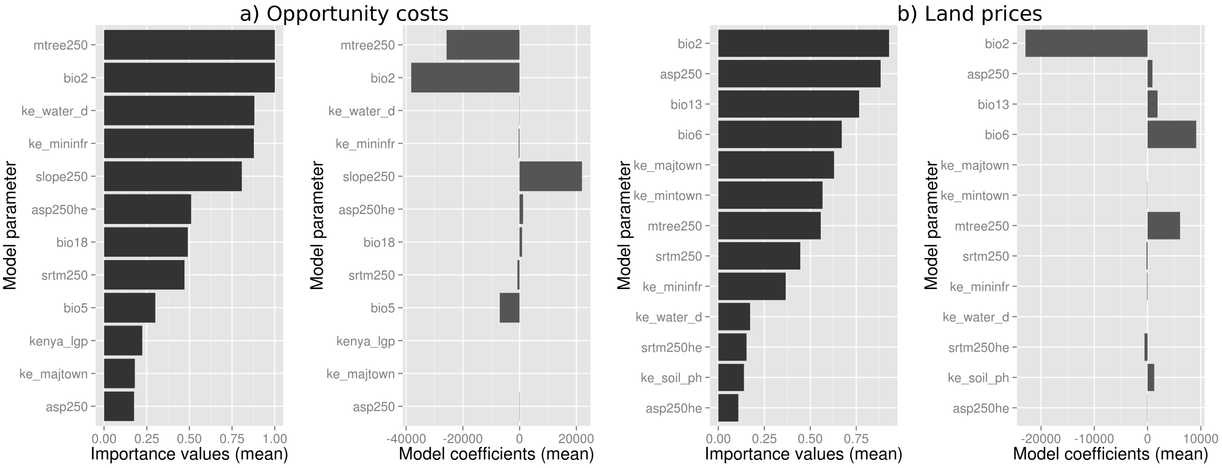


Figure A.8.2 – Model selection results for opportunity cost (a) and land price (b) GLMs. Importance values and model coefficients for each predictor of opportunity costs (a, left) and land prices (b, right) from the model selection and averaging procedure. Note that coefficient magnitudes express the change in raw response value per logarithmic change in predictor. [mtree250 = percentage tree cover; srtm250 = elevation; kenya_lgp = length of growing period; ke_majtown = distance to nearest major town; asp250 = aspect; asp250he = aspect heterogeneity [s.d.(asp250]; ke_water_d = distance to nearest water body; ke_mininfr = distance to nearest minor infrastructure (transport networks and settlements); ke_soil_ph = soil pH; bio2 = mean diurnal temperature range ; bio5 = max temperature of warmest month; bio13 = precipitation of wettest month; bio18 = precipitation of warmest quarter]

land prices ($n = 227$, $p = 0.002$, $R^2 = 0.043$), which has strong implications for equity and the participation of poorer households (see discussion in the main text).

8.3.3 Spatial modelling of opportunity costs and land prices

Model selection algorithm

We used an identical model selection algorithm to spatially map both opportunity cost and land price data. For opportunity costs, we excluded all households with negative values to avoid including data resulting from crop failure or abandonment, and because the degree of negativity is not informative. To correct for this in the final models, we adjusted predicted values downwards to account for the probability of crop failure (i.e. the proportion of negative or zero values). We used generalized linear models (GLMs) to elucidate the most important determinants of agricultural returns and land prices. GLMs integrate both categorical and continuous predictors and are robust to deviations from normality in response error (Venables & Ripley, 2002a).

We used a range of local household-specific and spatial environmental variables as predictors in the statistical models to explain variations in opportunity costs (see Table A.8.2 for a complete list and Fig. A.8.2 for most important variables). We justified the inclusion of each predictor in our model selections based on a hypothesized mechanism and effect direction on agricultural returns (e.g. distance to major infrastructure as a proxy for

Predictor	Description
Distance to major infrastructure [m]	distance to nearest main road or large town; data from WRI (2007)
Distance to minor infrastructure [m]	distance to nearest road (minor and major), town, village or trading centre; data from WRI (2007)
Distance to nearest settlement [m]	distance to nearest town, village or trading centre; data from WRI (2007)
Distance to nearest major town [m]	distance to nearest major town; data from WRI (2007)
Distance to nearest water body [m]	distance to nearest river, lake or wetland; data from WRI (2007)
Distance to nearest Protected Area [m]	distance to nearest protected area boundary; based on data from the World Database on Protected Areas 2010 release, all IUCN categories (UNEP-WCMC, 2010)
Soil pH	soil pH values from International Livestock Resource Institute (www.ilri.org).
Length of growing period [d]	number of growing days per year from International Livestock Resource Institute (www.ilri.org).
Slope [%]	modelled at 90m resolution and rescaled to 250 m using SRTM90 dataset.
Elevation [masl]	modelled at 90m resolution and rescaled to 250 m using SRTM90 dataset.
Topographic heterogeneity	standard deviation of 9 x 9 cell moving window. Modelled at 90m resolution and rescaled to 250 m resolution using the SRTM90 dataset.
Aspect [°]	modelled at 90m resolution and rescaled to 250 m using SRTM90 dataset.
Aspect heterogeneity	standard deviation of 9 x 9 cell moving window. Modelled at 90m resolution and rescaled to 250 m resolution using the SRTM90 dataset.
Bioclimatic variables (BIO1–19)	19 aggregate bioclimatic variables for temperature and rainfall from the WorldClim 1 km resolution global dataset (Hijmans *et al.* , 2005), consisting of 1: Annual Mean Temperature [°C], 2: Mean Diurnal Range [°C], 3: Isothermality [%], 4: Temperature Seasonality [°C*100], 5: Max Temperature of Warmest Month [°C], 6: Min Temperature of Coldest Month [°C], 7: Temperature Annual Range [°C], 8: Mean Temperature of Wettest Quarter [°C], 9: Mean Temperature of Driest Quarter [°C], 10: Mean Temperature of Warmest Quarter [°C], 11: Mean Temperature of Coldest Quarter [°C], 12: Annual Precipitation [mm], 13: Precipitation of Wettest Month [mm], 14: Precipitation of Driest Month [mm], 15: Precipitation Seasonality [coeff. of variation of monthly precipitation], 16: Precipitation of Wettest Quarter [mm], 17: Precipitation of Driest Quarter [mm], 18: Precipitation of Warmest Quarter [mm], 19: Precipitation of Coldest Quarter [mm]
Tree cover [%]	Based on MODIS Vegetation Continuous Field yearly dataset for 2010 (ftp://ftp.glcf.umd.edu/modis/VCF/Collection_5).

Table A.8.2 – Description of predictors used in the spatial models for both agricultural net returns and stated land prices.

market access). Although we did not posit direct relationships between certain (mainly climatic) predictors and land prices, we included all variables from the opportunity costs models because differences in agricultural potential influence capitalized land value (i.e. the expected stream of agricultural net revenues from the land over a discounted future time horizon), and therefore partly determine current land prices. For selecting variables to included in a final model, we used an information-theoretic approach based on Akaike's Information Criterion corrected for small samples (AICc) (Burnham *et al.* , 2011). A model selection and averaging process was used to randomly recombine predictors in new model configurations, and derive model weights, w, based on the negative exponent of the information distance between model i and the best model encountered (Calcagno & de Mazancourt, 2010), formally:

$$w_i = exp(-(AIC_i - AIC_{best})) \tag{8.3}$$

Importance values (ranging 0–1) were generated for each predictor based on the number of models in which they were chosen, weighed by the AIC weight, w_i, of the model. After an initial selection run, we removed strongly correlated predictors (correlation coefficient > 0.7) by choosing the variable with a higher importance value (i.e. more frequently chosen during predictor selection). We then repeated the selection procedure with the remaining uncorrelated predictors to identify the final model predictors (results of importance ranking are shown in Figure A.8.2). This selection procedure avoids the problems of automated step-wise selection (e.g. inflated degrees of freedom, pseudo-optima), but is less computationally intense than an exhaustive search of all potential predictor combinations (Calcagno & de Mazancourt, 2010). To investigate model fit and prediction error, we checked the structural goodness of fit of the best model (i.e. the "deviance explained", the proportion of null-model deviance that can be explained by the model) and also calculated total predicted model error using leave-one-out cross-validation.

The results indicated tree cover was an important predictor, negatively related opportunity costs. This is appropriate for occupied agricultural lands where shading may cause crop losses (and the fact that we ignored forestry returns, thus missing potential compensating effects), but extrapolating this model to the potential returns of intact forest areas and arid, but otherwise vegetation free, lowland regions would lead to biased results. To address this, we constructed a separate model omitting tree cover and used this to predict opportunity costs in areas of the landscape with tree cover percentage beyond the range of our survey data ($< 5\%$ and $> 60\%$ tree cover). The model parameters for the final spatial models for both opportunity costs and land prices are shown in Table A.8.3.

Predictor	Coefficient	Std. Error	DF	t-value	p-value
Opportunity costs (DE = 18.25%)					
(Intercept)	21.368974	2.7638559	175	7.73158	<0.0001
Tree cover [%]	-0.046378	0.0110233	175	-4.207239	<0.0001
Aspect heterogeneity [s.d. of aspect]	0.003336	0.0022265	175	1.498206	0.1359
Mean diurnal temperature range [°C]	-0.055853	0.0189632	175	-2.945338	0.0037
Distance to nearest water body [m]	0.000201	0.000053	175	3.788505	0.0002
Distance to nearest road or town [m]	-0.001271	0.000519	175	-2.449022	0.0153
Slope [%]	0.054738	0.0137542	175	3.979704	0.0001
Land prices (DE = 47.4%)					
(Intercept)	15.05861	0.716952	215	21.003651	<0.0001
Aspect [°]	0.000786	0.0003162	215	2.485816	0.0137
Mean diurnal temperature range [°C]	-0.013846	0.0046347	215	-2.987462	0.0031
Min Temperature of Coldest Month [°C]	0.009905	0.0019228	215	5.151493	<0.0001
Distance to minor town [m]	-0.000059	0.000014	215	-4.226877	<0.0001
Tree cover [%]	0.01279	0.0034135	215	3.74704	0.0002

Table A.8.3 – Model parameters for the opportunity cost (top) and land price (bottom) GLMs. DE = Deviance Explained, DF = degrees of freedom.

Accounting for spatial autocorrelation

Environmental and economic data often suffer from spatial autocorrelation due to non-independence amongst nearby sites (Beal *et al.* , 2010). We tested for spatial autocorrelation using Moran's test and by visualizing spatial lag using semi-variograms. We corrected for autocorrelation in the final models using the Generalized Least Squares (GLS) method, fit using penalized quasi-likelihood with correlation structure derived from visually interpreting semi-variograms (Dormann *et al.* , 2007; Beal *et al.* , 2010; Venables & Ripley, 2002b).

Literature opportunity cost data

We converted all literature values to 2010 dollars at an exchange rate of \$1 = KSh83, correcting for inflation using the Consumer Price Index from the Kenyan Bureau of Statistics (http://www.knbs.or.ke/). Where possible, we differentiated between small- and large-scale farms based on the criteria of Norton-Griffiths & Southey (1993). We spatially mapped these values by attributing them to a land cover classification system consisting of a cross between a simplified version of the Africover land cover product (FAO Africover, 2004) and the agroecological zones of Kenya (1: per-humid, 2: humid, 3: semi-humid, 4: transitional, 5: semi-arid, 6: arid and 7: per-arid; WRI, 2007).

To do this, we developed a land use map combining the FAO Africover land cover product with a map of the 7 FAO Agroecological Zones of Kenya. We classified the Africover classes into intensive and extensive land use types, and applied average values

AEZ	Production system	Literature OC cost estimate [$/ha/y]					Districts/localities	References
		Middle	High	Min	Max	N		
1. Humid*	crops, small-scale	1’639	2’459	104	4’323	4	Mau forest, national surveys	1,2
	crops, large-scale	2’310	3’466	297	4’323	2	Mau forest, national surveys	1,2
	(agro)forestry, woodlots	133	200	65	202	2	Mau forest, public data	1,3
	plantation, roundwood	494	742	202	847	5	Mau forest, public data	1,3
2. Sub-humid	crops, small-scale	1’023	1’712	188	4’323	13	Kakamega, national surveys	1,4
	crops, large-scale	3’784	5’899	278	16’965	9	National surveys	2
	livestock, zero and semi-zero	731	847	–	–	1	National surveys	2
	(agro)forestry, woodlots	96	144	21	21	3	Meru, Mau forest, public data	1,3,6
	plantation, roundwood	432	647	212	847	4	Mau forest, public data	1,3
3. Semi-humid	crops, small-scale	670	1’004	248	1’174	18	Mbeere, national surveys	2,5
	crops, large-scale	1’917	2’875	286	16’965	13	National surveys	2
	livestock, zero and semi-zero	387	580	–	–	1	National surveys	2
	livestock, ranching	29	44	–	–	1	National surveys	2
	(agro)forestry, woodlots	95	143	19	29	3	Meru, Mau forest, public data	1,3,6
	plantation, roundwood	432	647	212	466	4	Mau forest, national surveys	1,3
4. Transitional	crops, small-scale	563	844	95	1’400	15	Golini-Mwalunganje, Mbeere, national surveys	5,10
	crops, large-scale	587	881	244	1’400	10	Golini-Mwalunganje, Mbeere, national surveys	2,5,10
	livestock, ranching	17	26	–	–	1	National surveys	2
	(agro)forestry, woodlots	4	6	3	7	12	Makueni, Kitui, Mbeere and Tharaka	7
5. Semi-arid	crops, small-scale	577	865	95	1’400	13	Mbeere and Kijiado districts	6,8
	crops, large-scale	732	1’097	284	1’400	6	Mbeere and Kijiado districts	6,8
	livestock, ranching	64	96	26	136	3	Kimana Sanctuary, Kijiado and Machakos	2,8,10
	(agro)forestry, woodlots	3	4	1	6	12	Makueni, Kitui, Mbeere and Tharaka	7
6. Arid	crops & livestock, mixed	164	246	–	–	1	Narok district	9
	livestock, ranching	7	10	–	–	1	Narok district	9

*Due to a lack of data, we assumed half the opportunity cost estimates of AEZ 2; 1 = Kipkoech *et al.* (2011), 2 = Norton-Griffiths & Southey (1993), 3 = Sedjo (2004), 4 = Boerner *et al.* (2009), 5 = Mucheru-Muna *et al.* (2010), 6 = Holding *et al.* (2006), 7 = Wekesa *et al.* (2012), 8 = Nkedianye *et al.* (2009), 9 = Norton-Griffiths (1996), 10 = Mburu *et al.* (2003)

Table A.8.4 – Opportunity cost data from the literature (inflation-corrected, presented in 2010 $). Sample size, N, refers to number of instances of a land use, not the number of studies.

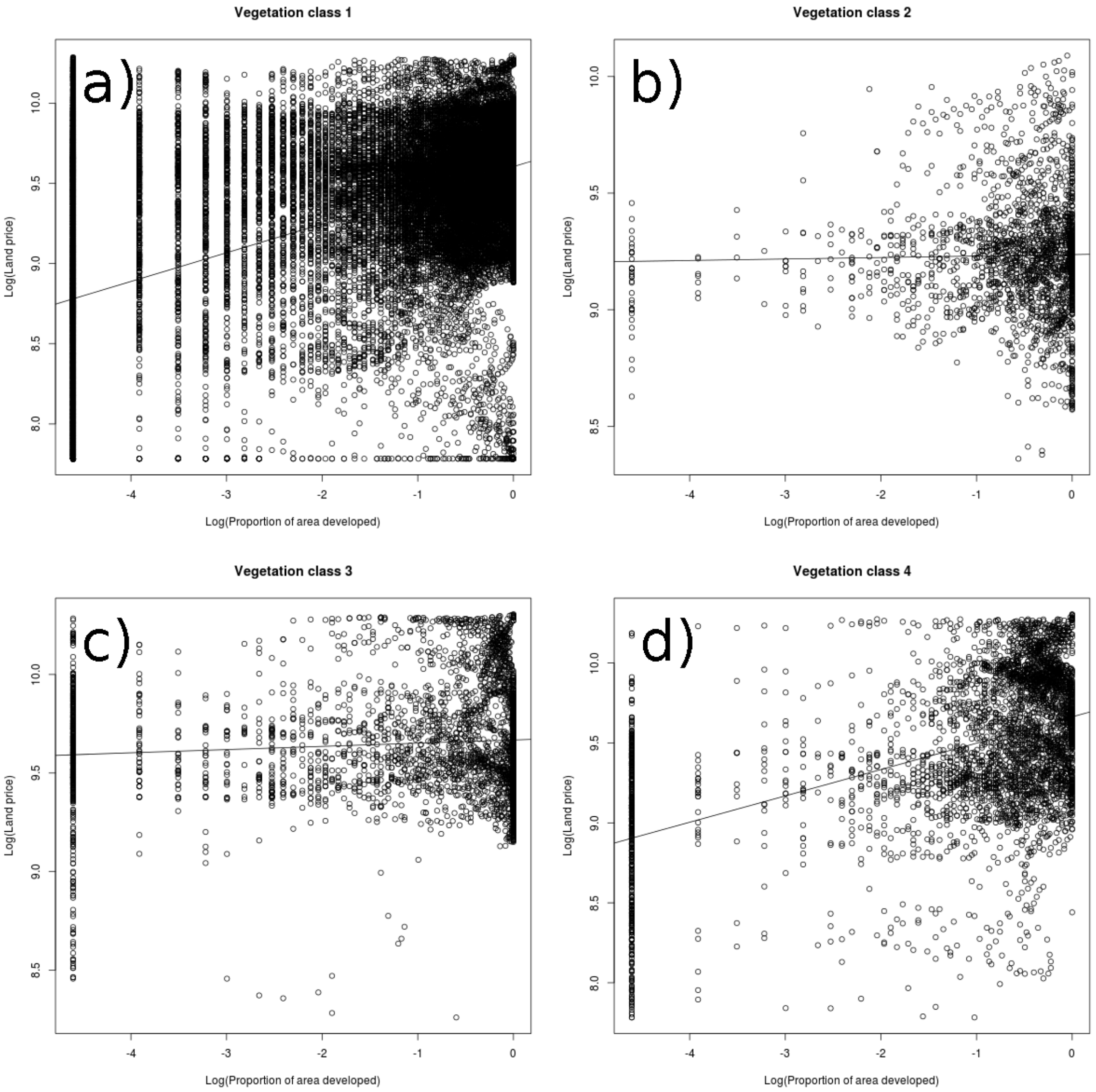


Figure A.8.3 – Linear regressions of land prices and the proportion of area developed within a 21 x 21 km neighbourhood for each vegetation class. Since classes 5–8 are adequately represented by public protected areas (i.e. were never selected for conservation intervention during the scenarios), only classes 1–4 are shown (a–d). The coefficients of these regressions were used to model price increases due to conservation land purchases or easements.

for literature opportunity costs per land cover class and agroecological zone. Finally, within each land cover class we weighted the average literature opportunity costs with a spatial layer of the length of growing period (i.e. adding spatial differentiation to the otherwise homogeneous literature data to reflect higher than average returns in areas with a longer growing period and vice-versa). For the final spatial models, we compared modelled and literature opportunity costs within each agroecological zone (to verify they were in the same range), and took the average value for further modelling. This combined the finer spatial variability of the modelled data with the added representation of the literature accounts (Fig. 4.1 in the main text).

Land market model

We developed a simple land market model to accommodate the change in prices caused by conservation purchases. The model was based on a simple log-log relationship between the amount of land developed within a 21 x 21 km grid, and the predicted land price. An separate relationship was established for each vegetation class (see section 8.3.1). Figure A.8.3 shows the relationship for selected vegetation class. The coefficient of the regression line describes the predicted increase in land prices per unit of land set aside for development or conservation (i.e. taken out of the market). We plotted separate relationships within the 8 vegetation classes. For simplicity in the scenarios, we assumed that price increases due to purchases are spread across the entire vegetation class, rather than using a neighbourhood effect (i.e. we assumed high substitutability of land within a vegetation class). We ignored the possibility that landowners who sell land or accept easements might migrate to urban areas, changing demand for urban services and living space rather than rural demand for land, as this was beyond the scope of our study.

Because the proportion of developed land was not used as a predictor in our land price model, there is no circularity in our approach to determine price increases as a function of the proportion of developed land. We applied this model to land prices after each round of conservation purchases in our scenarios (see 8.3.5). To be conservative, and to accommodate for potential amenity effects (e.g. increasing the value of development land in surrounding areas due to the presence of the reserve and associated nature-based amenities), we doubled the price increase in our baseline scenario. For the development scenario, we did not double the price increase, assuming that the development programme offsets the negative supply effect (e.g. through setting aside a small fraction of land for human settlement or improving land productivity, thus making it less likely that the conservation purchase would lead to *new* competition in the land market).

8.3.4 Organization interviews

Organization profiles

To collect data on conservation management costs, we interviewed four regional conservation organizations, two from rangeland areas, two from montane areas. Both rangeland conservancies (Lewa and Il Ngwesi) have characteristic large reserves (21'500 ha and 9'900 ha, respectively), with well-established security, wildlife monitoring, ecotourism, eco-enterprise and socio-economic development programmes. In contrast, the montane organizations represent local "Site Support Groups" of Nature Kenya, the national Affiliate of BirdLife International (Birdlife International, 2010), and act primarily through decentralized and indirect ICDPs with member households. Each organization is briefly profiled below.

Lewa Conservancy. The Lewa Conservancy covers an area of 25'100 hectares of privately owned rangeland and operates as a strict nature sanctuary with limited ecotourism activities (e.g. an upper limit of 118 visitor nights per month for each of 5 lodges). Operational since 1995, the conservancy is currently in the process of purchasing all land from its private owners and holding it in trust in perpetuity, supported by a newly created conservation endowment. Within Lewa's conservation standards is a mandate for development activities in the buffer zone of the reserve in order to facilitate benefit sharing with surrounding communities (e.g. roughly $1 million was spent on development-related activities in 2011).

Il Ngwesi Community Trust (Il Ngwesi). The Il Ngwesi Community Trust (lying adjacent to Lewa and also established in 1995) covers 13'200 hectares of rangeland and operates as a community ranch turned wildlife sanctuary, using ecotourism revenues for development activities within the conservancy. Roughly 25% of the land (3'300 ha) is designated for settlement and ranching activities with the remaining 75% (9'900 ha) set aside for wildlife protection and ecotourism. The reserve receives significant support from Lewa in terms of expertise, training, equipment and finances. As part of the Northern Rangeland Trust, Il Ngwesi is primarily funded through donations, supplemented with ranching, ecotourism and eco-enterprise ventures.

Mount Kenya Biodiversity and Conservation Group (MtKeBio). MtKeBio was established in 1998 as a Birdlife International local Site Support Group (SSG; Birdlife International, 2010). The group conducts ICDP activities around the buffer zone of Mount Kenya Forest Reserve, including direct support for sustainable alternative livelihoods (e.g. bee keeping, ecotourism capacity building, training) and efficient natural resource use (e.g. efficient wood burning stoves, reforestation). Land purchase and reserve establishment is also an explicit strategy, but is limited by budget constraints and high land prices (currently only 6.5 ha of closed bush are protected at the organizations office). MtKeBio currently has about 2'500 member households with an average farm size of 4 acres (1.62 ha). To account for stated funding shortfalls, we included hypothetical costs of a second resource centre and sanctuary with a quadrupling of yearly project funding (to KSh 60 million/y).

Friends of Kinangop Plateau (FoKP). FoKP was also established as an SSG in 1998 in response to growing agricultural intensification on the Kinangop Plateau, a 40'000 hectare expanse of montane tussock grassland on the western aspect of the Aberdares Range. The group focuses on promoting traditional extensive management of tussock grassland for conservation, particularly targeting the endangered and locally endemic Sharpe's Longclaw, *Macronyx sharpei* (Muchai *et al.* , 2001; Ndang'ang'a *et al.* , 2002). The group currently has two existing reserves with a further two in planning, totalling 75 hectares, and engages roughly 400 households with an average land holding of 6 ha per household.

Cost categories

We collected information on conservation production and transaction costs from the four aforementioned conservation organizations during semi-structured interviews. We differentiated between investment and operational phases. Production costs were categorized into land value (*investment phase only*, valued using our predictive model described above); habitat rehabilitation (*investment phase only*, covering vegetation restoration and enhancement, installation of water intakes and troughs to service wildlife, and a flat budget of $3 million for wildlife translocations for Lewa and Il Ngwesi based on recent operations at the Samburu Reserve; Geoffrey Chege, Lewa Chief Conservation Officer, pers. comm.); infrastructure and material (*both phases*, represent the value of all non-land assets including buildings, vehicles, fence construction, office furnishings etc. during investment, and representing upkeep and replacement costs during the operational phase); transport and operations (*operational phase only*, consisting cost of anti-poaching patrols, site visits, and material transport); administration (*operational phase only*, covering senior salaries and administration fees); labour (*both phases*, consisting non-administrative salaries such as casual labour, guard salaries, and training); development-related activities (*operational phase only*, based on development budgets and including all ICDP activities by the organizations); opportunity costs (*operational phase only*, valued using our predictive model described above).

Transaction costs were categorized into initial surveys (*investment phase only*, covering expenses of initial planning, concept development and information gathering on site suitability etc.) and negotiations (*operational phase only*, consisting negotiation time, valued for material costs of amenities, equipment and transport etc., and labour time by participants). We valued generic labour time according to salaries for agricultural labour (KSh 300 per day), with a higher rate for community leaders (KSh 600 per day), and administrative positions in local organizations (KSh 20'000 per month) and a fixed salary for conservation administrators (KSh 40'000 per month). These figures were adjusted as necessary based on interview responses. A breakdown of the results of our surveys is provided in the Table A.8.5.

Because the majority of the conservation activities of the montane organizations (MtKeBio and FoKP) are indirect (see organization profiles below), costs per unit of conservation outcome (e.g. \$ per hectare protected) are difficult to quantify (Kiss, 2004; Ferraro & Pattanayak, 2006)). To work around this, we isolated management costs associated with the relatively small, strict protected reserves currently under the management of these organizations from the remaining ICDP costs. Unfortunately, due to the small size of the reserves (ranging 6.5 to 20 ha and totalling 88 hectares of mostly demonstrative conservation land), we likely overestimated costs by omitting the effect of scale, and therefore our cost estimates for forest and montane grassland areas are likely in the upper range.

We split the remaining management costs between pure ICDP expenses, and more general costs involved in monitoring and negotiating conservation and development projects with households. We used the latter as a surrogate for potential transaction costs of a hypothetical PES scheme, after carefully considering each item included. The hypothetical PES scheme thus assumes each member household receive direct payments to rehabilitate native vegetation on a portion of their land. Thus we transferred a selection of transaction costs from an existing ICDP programme to a hypothetical PES scheme based on scenario building. For the LPE strategy, actual transaction cost data were available from previous conservation land purchases.

We could not estimate compensation costs due to wildlife damages, which would have required in-depth interview data with affected households. Finally, because the budget information we received was often aggregated, our cost categories contain a degree overlap and should be interpreted with care (e.g. the cost of habitat rehabilitation or fence construction aggregated all relevant costs, including labour, material and transport).

Management cost results

Our interviews revealed variable costs across the four organizations (Table A.8.5), with similar values within ecosystem type (rangeland and montane). Bear in mind that the figures shown in Table A.8.5 are hypothetical, and are composed of both existing and extrapolated funding needs. We separated formal labour costs (labour time to establish projects by initiators and other actors that was mostly paid) from informal costs made up of the value of negotiation time for community members and landowners (in brackets in the Table A.8.5). Per-hectare transaction costs across both phases were generally low for the rangeland conservancies (Lewa and Il Ngwesi), due in part to their large size and lower negotiation frequency, but informal costs (born by communities and landowners) often equalled or exceeded formal costs (Mburu *et al.* , 2003). Transaction costs were much higher for the montane organizations, due to proportionally larger reported costs in

Category	**Cost [$ in 2010]**			
	MtKeBio	**FoKP**	**Lewa**	**Il Ngwesi**
Reserve/PES area (ha)	1’013	675	25’100	9’470
Investment phase				
Transaction costs	29’217	16’268	46’276	72’094
Per unit area [$/ha]	28.9	24.1	1.8 [4.4]	7.6 [6.5]
Production costs				
Land value	150’634	993’167	491’743’700	100’078’013
Habitat rehabilitation	3’765’710	14’202	3’518’145	3’402’978
Infrastructure and material	391’437	42’746	5’424’445	996’108
Labour	24’942	3’974	229’340	107’925
Subtotal	4’332’722	1’054’090	500’915’631	104’585’025
Per unit area [$/ha]	4’279	1’562	19’957	11’044
Operational phase				
Transaction costs	63’134	9’621	55’557	7’843
Per unit area [$/ha/y]	62.4 [63.5]	14.3 [18.1]	2.2 [0.3]	0.8 [1.4]
Production costs				
Infrastructure and material	3’451	784	357’330	30’820
Transport and operations	5’386	1’882	730’993	11’765
Administration	202’228	11’700	2’240’265	198’045
Labour	5’961	6’353	275’208	164’805
Development-related activities	771’265	6’719	1’006’880	2’179’503
Subtotal	988’291	27’439	4’610’676	2’584’938
Per unit area [$/ha/y]	976	41	184	273
Opportunity costs	1’327’920	2’419’800	8’659’500	1’250’040
Per unit area [$/ha/y]	1’312	3’585	345	132
Total operating costs [$/ha/y]	2’350	3’640	531	406

Table A.8.5 – Management cost estimates for the four conservation organizations, split into investment or operational phase, and disaggregated to per-hectare values presented. Figures are a mix of actual and extrapolated data based on estimates of funding shortfalls. Transaction costs in square brackets represents unpaid labour time during negotiations, not included in totals.

establishing reserves and activities (searching for suitable sites, interacting with various actors) and a higher frequency of stakeholder decision-making and conflict resolution meetings, due to higher population densities and more intense and dynamic land and forest use trends.

Production costs were more variable, with investment costs for the Lewa conservancy ($19’957 per ha) dominated by the land value (ca. $491 million) of the extensive reserve. Both montane organizations had relatively low production costs in the investment phase ($4’279 and $1’562 per ha), due to modest infrastructure needs and minimal land purchases. However, opportunity costs are predicted to be much higher than the rangeland organizations, given the higher agricultural suitability of montane areas. For forest habitat, we did not have estimates of habitat restoration costs from the conservation organizations. Therefore, we used an estimate of $3’695 ha^{-1}taken from a global average of Nesshöver *et al.* (2011) (justified given the relatively close match between our average estimates from rangeland restoration of \$249 ha^{-1} compared to \$260 ha^{-1} in Nesshöver *et al.* , 2011).

8.3.5 Scenario analysis

Wunder (2005; 2006) and Engel *et al.* (2008) consider conservation easements as a form of PES, given that both entail a voluntary contract between buyer (*beneficiary*) and seller (*trustee*), retention of title deed, and conditionality. However, Hanley *et al.* (2012) and Miteva *et al.* (2012) differentiate based on the time frames involved. PES are time-constrained contracts subject to renegotiation, whereas easements are generally "in perpetuity" and non-negotiable after initial agreement (but use rights can be bought back, and easements can be sold on, by the trustee). In this study, we take the latter view, and assume that purchases and easements can be grouped together as long-term, inflexible tools, distinct from short-term, flexible PES contracts. Therefore in our scenarios, we grouped easements and purchases together (LPE) as a single strategy and compared this to the PES approach. We assumed that landowners engage in PES contracts if they exceed their opportunity costs (Engel *et al.* , 2008) and will sell land at prices reflecting both the capitalized value of land (i.e. factoring in perceptions of discounted future cash flows; Naidoo & Adamowicz, 2006) and demand for residential and commercial development (i.e. disconnected from agricultural returns). We also assumed that the value of an easement on all major use rights is approximated by the price of land.

For the 30 year future scenarios, we assumed that all land outside public protected areas was unprotected, and developed a targeting system that aims for 25% conservation coverage of each vegetation class. Within each class, criteria were set to target areas of lowest economic cost subject to the following constraints: only pixels of within the upper 10 percentile of mammal species diversity (using data from a recent IUCN Global Mammal Assessment; Rondinini *et al.* , 2011a), higher-than-average proximity to a public protected area, and lower-then-average development intensity (based on the proportion of a 1 km cell occupied by a "developed" land cover class—agriculture, pasture, infrastructure, urban, or plantation). The mammal diversity threshold reduced to higher-than-average after spending the initial investment budget, due to a lack of suitable combinations of criteria.

For the LPE scenario, the investment stage consisted of purchasing land iteratively, in steps of 1% towards the target coverage of 25% per vegetation class. Land prices adapted after each purchase round, and targeting criteria were recalculated. This process continued until the total investment budget was exhausted (or until yearly operating costs reach the annual budget, in which case new conservation investments stopped). For PES, the process was identical, except that the algorithm used opportunity costs instead

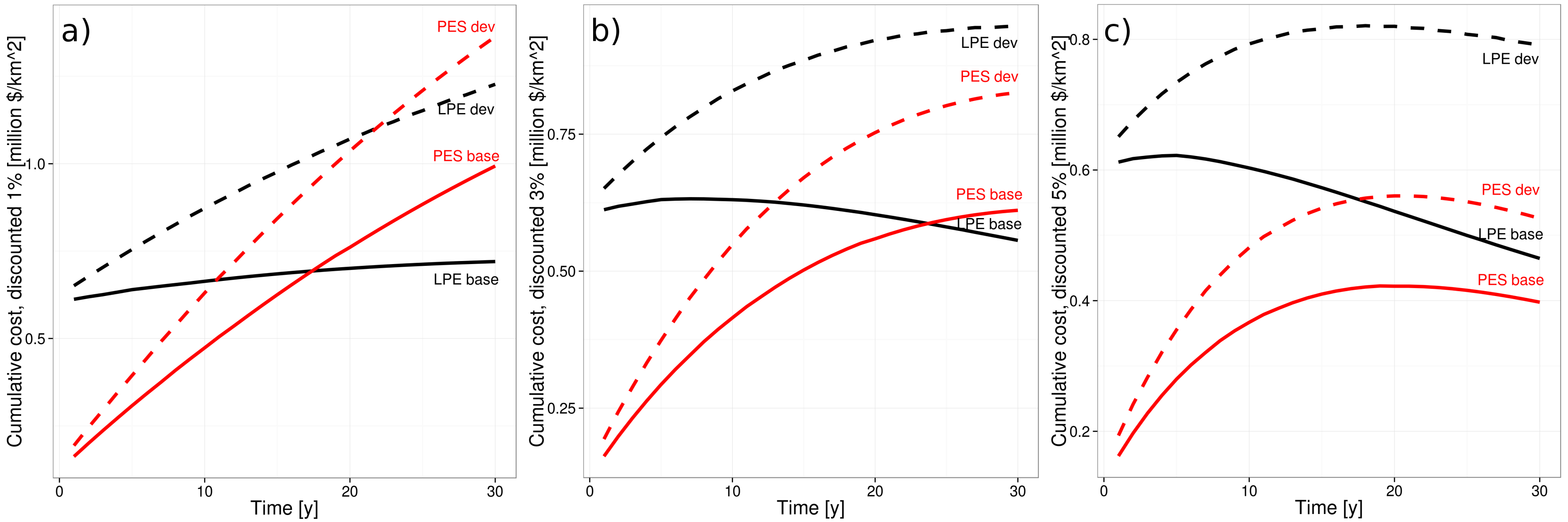


Figure A.8.4 – Scenario results of cost-effectiveness over a 30 year time horizon, with discounted future costs. Illustrates changes in cumulative costs per unit area discounted at 1% (a, left), 3% (b, middle) and 5% (c, right).

Cost category [unit]	Closed forest (6)	Sparse or degraded forest (3,4)	Montane grassland (2)	Lower rangeland (1)
Investment phase				
Transaction costs [$/ha]	28.7	28.7	24.1	4.7
Land value [$/ha]	modelled	modelled	modelled	modelled
LP processing fee [% land value]	25%	25%	25%	25%
Habitat rehabilitation [$/ha]	923.8	3’695	21	249.8
Production costs – investment [$/ha]	384	384	63	160.6
Operational phase				
LP transaction costs [$/ha/y]	2.5	2.5	7.0	0.5
PES transaction costs [$/ha/y]	62.0	62.0	14.3	1.5
Production costs – upkeep [$/ha/y]	23.8	23.8	27.3	93.2
PES OC [$/ha/y]	modelled	modelled	modelled	modelled
PES processing fee [% of OC]	25%	25%	25%	25%
Development programme [$/ha/y]	567.7	567.7	6.6	132.3
Development processing fee [$/ha/y]	189.2	189.2	3.4	2.8

Table A.8.6 – Per-hectare cost/benefit estimates used for spatial mapping to vegetation classes (bracketed numbers in header). Per-hectare production costs differ from Table A.8.5 because we separated out habitat rehabilitation costs, opportunity costs/land prices and development programme costs, which are presented individually. Higher altitude classes (5, 7, 8) were omitted, as they were adequately covered by public protected areas. Entry “modelled” refers to spatially modelled datasets based on empirical observations (see main text, Section 4.2.2 and Appendix 8.3.3). LP = land purchase, PES = payments for ecosystem services, OC = opportunity costs.

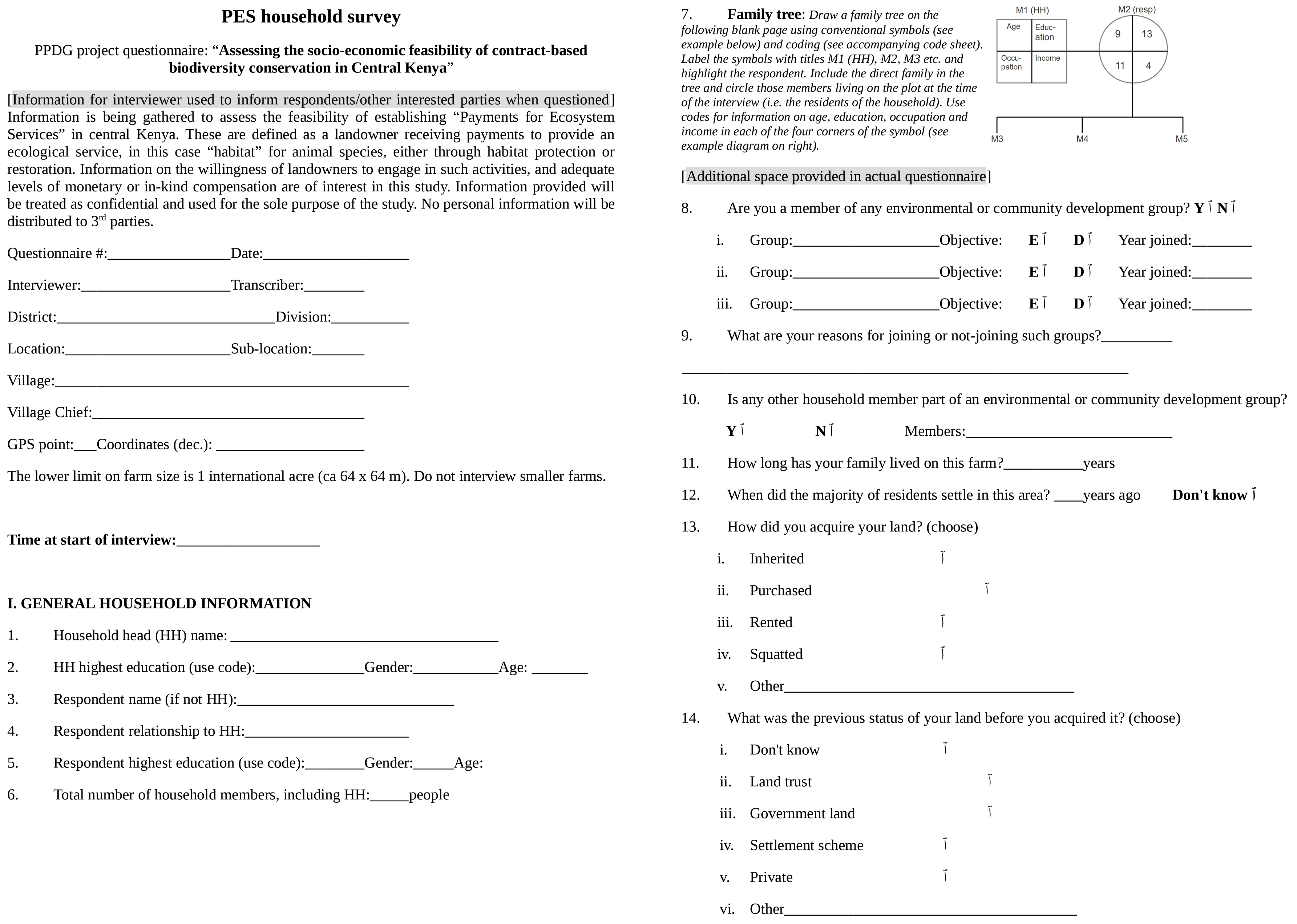
of land prices, no market feedbacks operated, and only part of the investment budget was spent (as the conservation organization must retain enough funds to cover initial PES contracts for the next thirty years). In the *operational phase*, a fixed annual budget was used either to meet existing PES contracts and operational costs, or expand the protection network to new contracts or purchases, with the associated investment costs. This process continued until the end of the modelling period.

For both strategies, converting a pixel to conservation management involves investing in habitat restoration, the capacity to monitor and manage the reserve area (resource centre, vehicles, security), infrastructure, equipment etc. (see Table 4.1 in the main text). The final per-hectare cost values used in the scenarios (investment and operational), per vegetation class, is provided in Table A.8.6. These values were derived either from the modelled land price or opportunity costs data ("modelled") or from the organization interviews.

To estimate the change in the proportion of species species represented in the reserve system at the end of the time period (referred to as "conservation coverage"), we used a species-area relationship (SAR) approach, formulated as:

$$C_{gain} = C_{t=30} - C_{t=0} = \left(\frac{A_{t=30}}{A_{total}}\right)^{z} - \left(\frac{A_{t=0}}{A_{total}}\right)^{z} \tag{8.4}$$

Where C_{gain} is the change in conservation coverage from the beginning ($C_{t=0}$) to the end ($C_{t=30}$) of the modelling period. A_{total} is the total potential habitat area in the region, $A_{t=30}$ is the area of the reserve system at the end of the modelling period, $A_{t=0}$ is the starting reserve area (only public PAs), and z is the slope of the species accumulation curve, set to 0.25, consistent with similar studies (e.g. Nelson *et al.* , 2009), and global average values for non-nested (independent) SARs (Drakare *et al.* , 2006).



PES household survey

PPDG project questionnaire: “**Assessing the socio-economic feasibility of contract-based biodiversity conservation in Central Kenya**”

[Information for interviewer used to inform respondents/other interested parties when questioned] Information is being gathered to assess the feasibility of establishing “Payments for Ecosystem Services” in central Kenya. These are defined as a landowner receiving payments to provide an ecological service, in this case “habitat” for animal species, either through habitat protection or restoration. Information on the willingness of landowners to engage in such activities, and adequate levels of monetary or in-kind compensation are of interest in this study. Information provided will be treated as confidential and used for the sole purpose of the study. No personal information will be distributed to 3rd parties.

Questionnaire #:________________Date:___________________

Interviewer:____________________Transcriber:________

District:_____________________________Division:__________

Location:______________________Sub-location:_______

Village:____

Village Chief:___________________________________

GPS point:___Coordinates (dec.): ___________________

The lower limit on farm size is 1 international acre (ca 64 x 64 m). Do not interview smaller farms.

Time at start of interview:___________________

I. GENERAL HOUSEHOLD INFORMATION

1. Household head (HH) name: ___________________________________
2. HH highest education (use code):______________Gender:___________Age: _______
3. Respondent name (if not HH):____________________________
4. Respondent relationship to HH:_____________________
5. Respondent highest education (use code):________Gender:_____Age:
6. Total number of household members, including HH:_____people

7. **Family tree**: *Draw a family tree on the following blank page using conventional symbols (see example below) and coding (see accompanying code sheet). Label the symbols with titles M1 (HH), M2, M3 etc. and highlight the respondent. Include the direct family in the tree and circle those members living on the plot at the time of the interview (i.e. the residents of the household). Use codes for information on age, education, occupation and income in each of the four corners of the symbol (see example diagram on right).*

[Additional space provided in actual questionnaire]

8. Are you a member of any environmental or community development group? **Y** ☐ **N** ☐
   i. Group:___________________Objective: **E** ☐ **D** ☐ Year joined:________
   ii. Group:___________________Objective: **E** ☐ **D** ☐ Year joined:________
   iii. Group:___________________Objective: **E** ☐ **D** ☐ Year joined:________

9. What are your reasons for joining or not-joining such groups?_________

____

10. Is any other household member part of an environmental or community development group?

    Y ☐ **N** ☐ Members:___________________________

11. How long has your family lived on this farm?__________years
12. When did the majority of residents settle in this area? ____years ago **Don't know** ☐
13. How did you acquire your land? (choose)
    i. Inherited ☐
    ii. Purchased ☐
    iii. Rented ☐
    iv. Squatted ☐
    v. Other______________________________________
14. What was the previous status of your land before you acquired it? (choose)
    i. Don't know ☐
    ii. Land trust ☐
    iii. Government land ☐
    iv. Settlement scheme ☐
    v. Private ☐
    vi. Other______________________________________

Figure A.8.5 – Example questionnaire used in the household surveys. Text in square brackets and shaded grey is used for explanatory purposes and was not included in the original form.

15. What land document do you have? (choose)
   i. None ☐
   ii. Letter of allotment ☐
   iii. Title deed ☐
   iv. Other____

16. Do you own additional land elsewhere? **Y** ☐ **N** ☐
   i. Where?____
   ii. Distance from here:__________km Size:________acres
   iii. How long have you owned it?__________________________
   iv. How did you acquire it (use options above)?______________
   v. Land document (use options above):____________________
   vi. Its general use(s):____________________________________

II. LAND USAGE

17. Total land area:________________________acres

18. Current land value:____________________Ksh/acre

19. What is the average acreage of land in this area?_________acres

20. What is the average wage for farm labour in this area?____Ksh/day

21. What is the average wage for an animal herder in this area?____________Ksh/day

22. What is the average wage for housekeeping labour in this area?_________Ksh/day

23. Do you employ housekeeping labour? **Y** ☐ **N** ☐ if yes,_ ______pers-days/month

24. What is the name of your local market, and how far away is it?
   Name:______________________________Distance:__________km

25. What is the name of the next larger market, and how far away is it?
   Name:______________________________Distance:__________km

26. Land area under:
   i. crop cultivation:____________acres
   ii. grazing:___________________acres
   iii. grass/hay production:________acres
   iv. farm forestry_________exotic acres________native acres
   v. buildings/habitation__________acres
   vi. other uses (specify)__________acres of____________________________________

27. Type of crops and production information ***per harvest*** (including timber and grass): *List each crop by name, note if the crop is inter-cropped with another and add the percentage area of each. Add the area planted, irrigated area, seasonal cost of farming inputs (e.g. chemical fertilizers, pesticides, tools), paid/unpaid labour (person-days of labour), estimated yield, share of yield consumed by household (% cons.) and sold at market (% sold), and selling price. Use the reverse of the sheet for any notes or calculations.*

Crop	**Intercropped (% area)**	**Area planted (acres)**	**Irrigated (acres)**	**Cost of inputs (Ksh/ harvest)**	**Paid labour (pers-days / harvest)**	**Unpaid labour (pers-days / harvest)**	**Yield (Kg/ harv.)**	**Kgs or bags cons.**	**Kgs or bags sold**	**Selling price (Ksh/Kg)**
Larger table provided in actual survey										

28. Type of livestock and production information ***per month or animal***: *List each livestock type and breed by name, enter the HH's estimates of herd size, water use, monthly cost of inputs (e.g. fodder, grain feeds, veterinary expenses), water usage, paid/unpaid herding or guard labour, whether the herd grazes on public land, share of animals consumed and sold, and market price per animal. Use the reverse of the sheet for any notes or calculations.*

Livestock type and breed	**# of animals**	**Water cons. (ltr/anim/ day)**	**Cost of inputs (Ksh/ month)**	**Paid labour (person -days/ month)**	**Unpaid labour (pers-days/ month)**	**Grazed on public land (y/n)?**	**# anim. cons. per year**	**# anim. sold per year**	**Selling price (Ksh/anim)**
Cattle									
Sheep									
Goats									
Chickens									
Donkeys									
Pigs									
Fish						N/A			
...									
Larger table provided in actual survey									

29. List any information on post-production costs and income of farm products ***per month***: *Include information on crops, livestock and additional animal or plant products.*

Costs						**Income**			
Product (crop, livestock or other)	**Process (Ksh/ month)**	**Treatment (Ksh/ month)**	**Pack (Ksh/ month)**	**Store (Ksh/ month)**	**Trans. (Ksh/ month)**	**Amount (Kg or units/ month)**	**units cons.**	**units sold**	**Selling price (Ksh/Kg or unit)**
Milk									
Eggs									
Furs/pelt									
Manure									
Larger table provided in actual survey									

Figure A.8.5 cont.

III. ENERGY USAGE

30. What is your main source of lighting
    - i. Kerosene: ______________ltrs/week
    - ii. Electric: ______________Ksh/month
    - iii. Solar: Installation costs of_____________Ksh___________years ago
    - iv. Other (specify with price):____

31. What is your main source of fuel?
    - i. Firewood: **From own farm** ☐ **From forest** ☐ **Purchased**_____Ksh/month
    - ii. Charcoal: **Own production** ☐ **Purchased** ☐____Ksh/month
    - iii. Gas: Cylinder size__Kgs, refilled every_____________which costs_____Ksh
    - iv. Kerosene: ______________ltrs/week
    - v. Electricity:**Y** ☐ **N** ☐
    - vi. Other (specify with price)_______________________________

IV. WATER USAGE

32. What is the price of water in this area: **Dont know** ☐ _______Ksh per________
33. What is your daily household water consumption:__(units) per_____________
34. Source of household water:**River** ☐**Well/Borehole** ☐ **Piped** ☐
35. Do you pay for your water? **Y** ☐ **N** ☐
36. How much water do you use for irrigation?_______________ltr/week
37. Type of farming irrigation:

    None ☐ **Furrow** ☐ **Drip** ☐ **Riverine** ☐ **Flooding** ☐ **Sprinkler** ☐

    Other____

38. Source of farming irrigation water:**River** ☐ **Well/Borehole** ☐ **Piped** ☐
39. Source of water for animals: **River** ☐ **Well/Borehole** ☐ **Piped** ☐
40. Do you own a fish pond? **Y** ☐ **N** ☐ Size___________m^3
41. How often do you refill the pond? ______________with___litres of water
42. Source of water for the fish pond? **River** ☐**Well/Borehole** ☐ **Piped** ☐

V. OFF-FARM INCOME, FINANCES AND PERCEPTIONS ON PES

43. Do you own or operate an off-farm business? **Y** ☐ **N** ☐
    - i. Type of business:______________________________________
    - ii. Location:____
    - iii. Monthly personal labour time:____________________person-days/month
    - iv. Monthly paid labour:___________________________person-days/month
    - v. Cost of labour:_________________________Ksh per person/day
    - vi. Monthly profit from business:____________________Ksh/month

44. Please provide information on any additional expenditures and investments in the past year:

Expenditures		**Investments/savings**	
Item	**Amount per...**	**Item**	**Amount per...**
Education and school fees		Land purchase	
Shopping (food, clothes...)		Land development	
Medical costs		Infrastructure purchase/upgrade	
Travel		Livestock purchase	
Leisure		Business investment	
Other...		Vehicle	
		Savings	
		Other...	
Larger table provided in actual survey			

Time at end of interview: ____________________________

Figure A.8.5 cont.

8.3.6 References

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8.4 Appendices for Chapter 5

Part A: Methods

8.4.1 Land use assessment framework in LCA

Assessing biodiversity impacts of land use in Life Cycle Assessment (LCA) adopts a two stage framework, reflecting two possible types of ecosystem damages: transformation impacts and occupation impacts (Koellner & Scholz, 2008; Mila i Canals *et al.* , 2007). These two impact types are depicted in Figure 8.4.1 by the trajectory of the yellow line, which indicates the change in biodiversity over time. After an initial land use change, such as a transformation from a natural to an anthropogenically-modified state, or from an extensive to an intensively managed state, biodiversity value (here represented by threat- and rarity-weighted species richness) of a reference situation, S_{ref}, is reduced to S_i. Assuming land is used for a specific period and for a specific purpose, such as crop production (the land use phase), natural regeneration of biodiversity is suppressed. At some hypothetical time in the future, land abandonment is assumed to take place, and biodiversity recovers to a level comparable to the pre-transformation state, S_{ref}. The duration of this regeneration is referred to as the regeneration or recovery phase, t_{reg}. In this context, impacts are considered to be fully reversible given a long enough time horizon. The final occupation (land use) and transformation impact (of the land use change) are given as the product of biodiversity loss, $\triangle S$, time, T, and area, A, affected (Koellner & Scholz, 2008; Mila i Canals *et al.* , 2007; Koellner *et al.* , 2013b). This is considered as the *Biodiversity Depletion Potential*, *BDP:*

$$BDP_{occ} = (S_{ref} - S_i) * A_{occ} * (t_1 - t_0) = \triangle S * A_{occ} * T_{occ} \tag{8.5}$$

$$BDP_{trans} = (S_{ref} - S_i) * A_{occ} * 0.5 * (t_2 - t_1) = \triangle S * A_{occ} * 0.5 * T_{reg} \tag{8.6}$$

The convention for life cycle inventory databases is to report land occupation of processes as area used over a certain time (blue area in Figure A.8.1). The inventory for occupation, I_{Occ}, is therefore:

$$I_{occ} = A_{occ,i} * T_{occ} \tag{8.7}$$

The biodiversity impact (or characterization factors in LCA terminology) of occupation, BI_{occ}, (blue arrow), is equal to the difference between the species richness of a reference state and that under a particular land use.

$$BI_{occ} = S_{ref} - S_i \tag{8.8}$$

For land transformation, the inventory, I_{Trans}, simply represents the area converted per process (purple arrow in Figure A.8.1):

$$I_{trans} = A_{trans} \tag{8.9}$$

The biodiversity impact of transformation BI_{trans} is given as (purple triangle in Figure A.8.1):

$$BI_{Trans} = (S_{ref} - S_i) * t_{reg} * 0.5 = BI_{occ} * t_{reg} * 0.5 \tag{8.10}$$

The time-lag involved in biodiversity regeneration is attributed to the initial transformation activity, rather than the occupation phase. This is due to the fact that even in the absence of occupation, a lag in diversity during recovery would result from a transformation activity (e.g. clear-cut logging of a forest and the subsequent recovery of native diversity). In some cases, the assumption of full recovery cannot be met. For example, a transformation may drive a species to extinction, biotic or abiotic conditions may be irreversibly modified (e.g. soil compaction, total topsoil loss), or the time horizon of recovery may be impractically long for meaningful prediction. In these cases, a permanent impact is said to occur (Koellner *et al.* , 2013b). We do not address permanent impacts in this study, restricting our analysis to occupation and transformation impacts. However, one approach for quantifying permanent impacts is given in de Baan *et al.* (2013b).

8.4.2 Local, relative method (***R-Local***)

In the method developed by de Baan *et al.* (2013a), local species richness of different types of land use were compared to (semi-)natural reference areas within the same biogeographic region. The relative reduction in local species richness was used as an indicator of impacts on biodiversity. Data was derived from a global literature review (GLOBIO3; Alkemade *et al.* , 2009), containing 195 publications providing 644 data points on different land use types and 254 data points on (semi-)natural reference situations from a total of nine out of 14 biomes. Because most data points came from tropical regions, de Baan *et al.* (2013a) complemented the dataset with national biodiversity monitoring data from

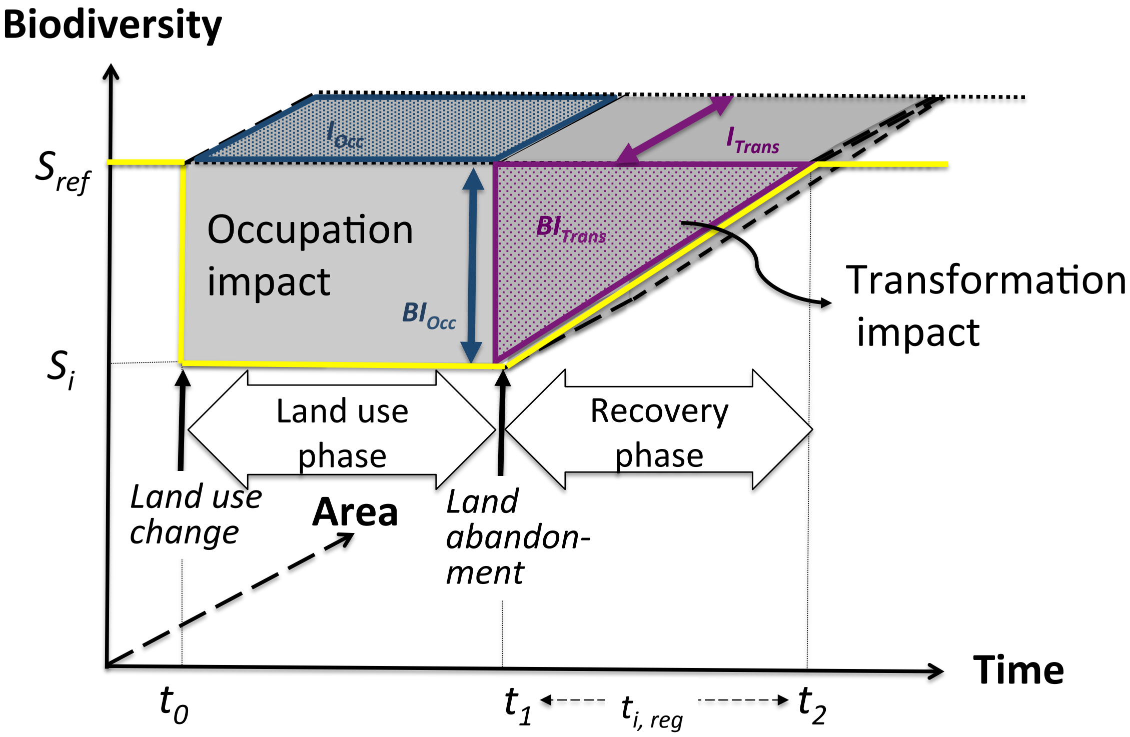

Figure A.8.1 – **Land use assessment framework in Life Cycle Assessment** (adapted from Koellner *et al.* , 2013b; Mila i Canals *et al.* , 2007).

Switzerland (temperate region) (BDM, 2004). In contrast to the GLOBIO3 model, which calculated reduction in mean species abundance of original species (MSA), de Baan *et al.* (2013a) used the GLOBIO3 database to calculate reduction in species richness, because this facilitated the inclusion of more studies (i.e. also studies that did not report species lists and abundance data). The GLOBIO3 dataset contains data on a broad range of taxonomic groups, while the BDM dataset contains only data on vascular plants, mollusks and mosses. In the present study, we used median biodiversity impacts (characterization factors) calculated per WWF biome (Olson *et al.* , 2001) over all taxonomic groups for the two land use types, annual and permanent crops. If less than five data points were available for one land use type in a biome, median global values were used instead. For permanent crops, only the biome Sub-Tropical Moist Forest had enough data to give a biome-specific biodiversity impact, all other biomes in the case study region (Savanna, Flooded Grassland, Montane Grassland, Desert) were assigned the median global value.

Biodiversity impacts of transformation (see Equation S6), not included in de Baan *et al.* (2013a), were quantified based on recovery times provided by (de Baan *et al.* , 2013b), based on the biodiversity recovery model developed by de Baan *et al.* (2013a) (see also section S5). To estimate transformation impacts, we used estimates of biome-specific recovery times per WWF realm (i.e. continent).

8.4.3 Regional, absolute method (*A-Regional*)

This method was developed by (de Baan *et al.* , 2013b) and estimates regional impacts of land use based on species-area relationships (Arrhenius, 1921; Koh & Ghazoul, 2010a). The method calculated the potential regional loss of species within WWF Ecoregions (Olson *et al.* , 2001) due to the historical conversion of habitat and allocated this loss to the different types of land use occurring in each Ecoregion. The matrix-calibrated species area-relationship developed by Koh & Ghazoul (2010a) was used to model regional species loss. This model accounts for the moderating effect of the habitat quality of human-modified land (i.e. the land use matrix) on species loss.

$$\frac{S_{new}}{S_{org}} = \left(\frac{A_{new}}{A_{org}}\right)^{z \sum_i^n p_i \sigma_i} \tag{8.11}$$

S is the regional species richness of a native, undisturbed (original) ecosystem ($\mathrm{S_{org}}$) and of the current land use mix ($\mathrm{S_{new}}$). $\mathrm{A_{org}}$ is the total area of the Ecoregion, while $\mathrm{A_{new}}$ is the remaining natural habitat area. The constant z indicates the species-accumulation rate observed in true island archipelagos (in de Baan *et al.* , 2013b, this was based on data presented by Drakare *et al.* , 2006). n is the total number of land use types, i, and p is the area share of each land use type within the total converted (i.e. non-natural) area. σ is a measure of the habitat quality of each type of land use, i. De Baan et al. (2013b) used the biodiversity impact scores (local relative change in species richness) from the previous study (de Baan *et al.* , 2013a) as an indication of habitat quality, σ, of different types of land use in different world regions. Species loss S_{loss} per Ecoregion was finally calculated as:

$$S_{loss} = S_{org} - S_{new} \tag{8.12}$$

Where S_{org} was derived from the WWF databases and S_{new} was calculated based on equation S7. We separately calculated regional species loss of endemic and non-endemic species. The potential loss of non-endemic species was used to calculate reversible biodiversity impacts of occupation (land use) and transformation (land use change). The number of potentially lost species which are endemic to one Ecoregion was used as a proxy of permanent land use impacts, as endemic species loss infers irreversible global extinction. However, in the present study, we only illustrate results based on reversible occupation and transformation impacts using non-endemic species.

Finally, de Baan *et al.* (2013b) allocated the regional species loss to the different types of land use present in each Ecoregion, j, based on the relative area share each land use type, i, occupies (p) and based on their repective habitat quality (σ). The allocation factor was calculated as:

$$a_{i.j} = \frac{p_{i,j}\sigma_{i,j}}{\sum_i^n p_{i,j}\sigma_{i,j}} \tag{8.13}$$

The biodiversity impact of land occupation (BI_{occ}, or characterization factor in LCA terminology) per land use type $\underline{i}$, region j, and taxonomic group, g, was calculated as a multiplication of the regional species loss by the allocation factor, divided by to total area of each type of land use $A_{i,j}$ per Ecoregion.

$$BI_{occ,reg,i,j,g} = \frac{S_{lost,nonend,j,g}a_{i,j}}{A_{i,j}} \tag{8.14}$$

Biodiversity impact of transformation were calculated based on equation S6. Biome-specific regeneration times t_{reg} were calculated per realm (i.e. continent) based on a meta-study of Curran *et al.* (2014) (see also Appendix 8.4.5).

The study of de Baan et al. (2013b) provides biodiversity impacts (BI) and their uncertainty (based on Monte Carlo simulations) for five taxonomic groups (mammals, birds, amphibians, reptiles, plants) and four land use types (agriculture, used forests, artificial area, pastures). In the present study, we only used the median values for mammals and the land use type "agriculture".

8.4.4 Calculation of life cycle inventory data

To estimate the land use change related to the production of 1kg of crop (tea, coffee, or tobacco), we adapted a three-step approach suggested by Milà i Canals *et al.* (2013). The approach is based on national statistics on land cover and harvested area per crop provided by the Food and Agriculture Organization of the United Nations (FAO) (FAOSTAT, 2013a,b). In a first step, the crop expansion rate per country was calculated over a 20 year time period (Table A.8.1).

Crop by country	**Average area 1991-1995 (ha)**	**Average area 2007-2011 (ha)**	**Area change, past 20 y (ha)**	**Expansion rate (%)**
Burundi				
Coffee, green	37000	19040	-17960	No LUC
Tea	6303	8900	2597	29.00%
Tobacco, unmanufactured	3951	1594	-2356	No LUC
Djibouti				
Coffee, green	No data	No data	No data	No data
Tea	No data	No data	No data	No data
Tobacco, unmanufactured	No data	No data	No data	No data
Eritrea				
Coffee, green	No data	No data	No data	No data
Tea	No data	No data	No data	No data
Tobacco, unmanufactured	No data	No data	No data	No data
Ethiopia				
Coffee, green	234118	397460	163342	41.00%
Tea	2751	7272	4520	62.00%
Tobacco, unmanufactured	5367	5641	274	5.00%
Kenya				
Coffee, green	157320	159600	2280	1.00%
Tea	106022	164991	58969	36.00%
Tobacco, unmanufactured	7106	18197	11090	61.00%
Rwanda				
Coffee, green	41130	34991	-6139	No LUC
Tea	10542	12937	2395	19.00%
Tobacco, unmanufactured	2846	4041	1195	30.00%
Somalia				
Coffee, green	No data	No data	No data	No data
Tea	No data	No data	No data	No data
Tobacco, unmanufactured	241	312	71	23.00%
Uganda				
Coffee, green	264200	308000	43800	14.00%
Tea	15600	24962	9362	38.00%
Tobacco, unmanufactured	7505	14996	7491	50.00%
United Republic of Tanzania				
Coffee, green	126992	119634	-7358	No LUC
Tea	18549	21483	2933	14.00%
Tobacco, unmanufactured	36120	74878	38758	52.00%

Table A.8.1 – Step 1, change in harvested area (in ha) of coffee, tea and tobacco over the past 20 years for all East African countries (data source: FAOSTAT, 2013b). No LUC = no land use change (crop area did not increase)

Land cover type by country	**Average area 1991–1995 (1000 ha)**	**Average area 2007–2011 (1000 ha)**	**Land cover change, past 20 y (1000 ha)**	**Expansion rate (% converted to forest)**
Burundi[1]				
Arable land	932	939	7	increase
Permanent crops	368	388	20	increase
Permanent meadow/pasture	835	916	81	increase
Other land	171	148	-23	decrease (21%)
Forest area	262	177	-85	decrease (79%)
Djibouti				
Arable land	1	2	1	increase
Permanent crops	No data	No data	No data	no data
Permanent meadow/pasture	1391	1700	309	increase
Other land	920	611	-310	decrease (100%)
Forest area	6	6	0	No LUC
Eritrea				
Arable land	458	682	224	increase
Permanent crops	2	2	0	No LUC
Permanent meadow/pasture	6945	6900	-45	decrease (20%)
Other land	1093	980	-113	decrease (50%)
Forest area	1603	1536	-67	decrease (30%)
Ethiopia				
Arable land	9957	13911	3953	increase
Permanent crops	547	985	438	increase
Permanent meadow/pasture	20000	20000	0	No LUC
Other land	54946	52668	-2278	decrease (48%)
Forest area	14550	12437	-2114	decrease (52%)
Kenya				
Arable land	5268	5420	152	increase
Permanent crops	480	610	130	increase
Permanent meadow/pasture	21300	21300	0	No LUC
Other land	26196	26106	-90	decrease (32%)
Forest area	3670	3478	-192	decrease (68%)
Rwanda				
Arable land	802	1190	388	increase
Permanent crops	275	250	-25	decrease (5%)
Permanent meadow/pasture	606	450	-156	decrease (32%)
Other land	458	152	-307	decrease (63%)
Forest area	326	425	99	increase

Table A.8.2 – Step 2, land cover change (in 1000 ha) for all East Africa countries over the past 20 years (data source: FAOSTAT, 2013b). No LUC = land cover area did not change.

Land cover type by country	**Average area 1991–1995 (1000 ha)**	**Average area 2007–2011 (1000 ha)**	**Land cover change, past 20 y (1000 ha)**	**Expansion rate (% converted to forest)**
Somalia				
Arable land	1027	1080	53	increase
Permanent crops	20	29	8	increase
Permanent meadow/pasture	43000	43000	0	No LUC
Other land	10635	11802	1167	increase
Forest area	8052	6824	-1228	decrease (100%)
Uganda				
Arable land	5044	6560	1516	increase
Permanent crops	1922	2200	278	increase
Permanent meadow/pasture	5112	5112	0	No LUC
Other land	3417	3033	-384	decrease (21%)
Forest area	4486	3076	-1410	decrease (79%)
United Republic of Tanzania				
Arable land	8901	11205	2305	increase
Permanent crops	1000	1680	680	increase
Permanent meadow/pasture	24000	24000	0	No LUC
Other land	14394	17864	3469	increase
Forest area	40285	33831	-6454	decrease (100%)

Table A.8.2 cont.

In a second step, the change in land cover types per country was assessed over the same time period (Table A.8.2). A crop was only considered to contribute to land use change if both the area of crop production (step 1) and the land cover type associated to this crop production (step 2) increased over this time period. Thereby, a pure shift from one crop to another (e.g. from maize to tobacco) was not considered as land use change. While the first two steps were done in line with Milà i Canals *et al.* (2013), we adapted the third step to better fit our aims. Milà i Canals *et al.* (2013) suggest to calculate the land use change rate per crop based on the change in land cover type. In our case study, this resulted in equal land transformation proportions per crop area for both coffee and tea (both grown on the land cover type “permanent crops”), although in some countries, such as Kenya, coffee only slightly increased in the past 20 years (+1%) while the tea cultivation area largely increased (+36%). Therefore, we adapted the last step of the approach and calculated the land use change rate per crop based on the change in crop area (and not on the change in land cover area, Table A.8.3).

In the present study, land use change was only considered from forest area (i.e. if the forest area decreased in the same 20 year period). The impacts of other land transformations (i.e. from pasture to annual crop) were not considered, as the methods to assess impacts on biodiversity (i.e., *R-Local, A-Regional* and *W-Local*) did not cover land transformations between different types of human use. Neither could we calculate the land conversion of savanna/grassland ecosystems, as the FAO database does not specify this land cover types: natural grasslands are contained within the land cover class “other”, which also contains very different land cover types, such as artificial areas. To minimize the influence of short-term fluctuations in land use, we used 5-year averages to calculate steps 1-3 (Milà i Canals *et al.* , 2013). The final amount of land use change per kg of crop was calculated by dividing the land use change rate by the crop yield (on a grid cell level).

To get a better understanding of the procedure, we illustrate the approach in the case of tea grown in Kenya. In step 1, we found that the harvested area of tea increased by 36%, from 106’022 ha (1991-1996) to 164’991 ha (2007-2011). In step 2, we considered the change of the land cover type “permament crops”, which increased by 21%, from 480’000 ha (1991-1996) to 610’000 ha (2007-2011). Because both step 1 and 2 showed an increase in crop and land cover area, respectively, we allocated land use change to tea production in Kenya. In step 3, we analysed which land use types decreased in the same time. Forest area decreased by 192’000 ha, while “other” land decreased by 90’000 ha in the 20 year period. Therefore, only 68% (=192’000/(192’000+90’000)) of the land use change of tea was associated with conversion of forest, the other 32% was not assessed. The final forest conversion rate associated with the growth in tea production was calculated as the multiplication in the land use change rate of tea (36%) and the share of forest converted (68%)[2]. This resulted in a land use change rate of 24% for tea in Kenya considering the land use changes over the past 20 years. This rate was finally divided by the yields of each grid cell (based on the data of Monfreda *et al.* , 2008) to obtain estimates of the amount of land use change associated with the production of 1 kg of crop in each grid cell within the borders of Kenya.

8.4.5 Calculation of recovery times of biodiversity

Biodiversity recovery times for the method *W-Local* were based on a spatial prediction of the models of Curran *et al.* (2014), who conducted a meta-analysis of the habitat restoration and secondary growth literature. The study analysed species checklist data from 39 comparative studies of secondary growth (SG) and old growth (OG) habitat from around the world. Changes in species similarity and assemblage composition were modelled as a

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[2] However, bear in mind that tea is adapted to environments that support forest growth, and therefore tea could be assumed to result exclusively in forest loss (i.e. “other” class ignored). However, the expansions in tea coverage could have occurred on already-transformed land represented in the “other” class, and therefore we were cautious in equating all crop expansion to forest loss.

Crop by country	Step 1 Crop expansion rate	Step 2 Increase in LC?	Step 3 LUC rate from forest	 LUC rate from "other"	Final LUC rate, forest to arable (tobacco) or permanent (coffee/tea)
Burundi			79.00%	21.00%	
Coffee, green	No increase	Yes			No LUC
Tea	29.00%	Yes			23.00%
Tobacco, unmanufactured	No increase	Yes			No LUC
Djibouti			0.00%	100.00%	
Coffee, green	No data	Yes			No LUC
Tea	No data	Yes			No LUC
Tobacco, unmanufactured	No data	Yes			No LUC
Eritrea			30.00%	50.00%	
Coffee, green	No data	No			No data
Tea	No data	No			No data
Tobacco, unmanufactured	No data	Yes			No data
Ethiopia			48.00%	52.00%	
Coffee, green	41.00%	Yes			20.00%
Tea	62.00%	Yes			30.00%
Tobacco, unmanufactured	5.00%	Yes			2.00%
Kenya			68.00%	32.00%	
Coffee, green	1.00%	Yes			1.00%
Tea	36.00%	Yes			24.00%
Tobacco, unmanufactured	61.00%	Yes			42.00%
Rwanda			0.00%	63.00%	
Coffee, green	No increase	No			No LUC
Tea	19.00%	No			No LUC
Tobacco, unmanufactured	30.00%	Yes			No LUC
Somalia			100.00%	0.00%	
Coffee, green	No data	Yes			No data
Tea	No data	Yes			No data
Tobacco, unmanufactured	23.00%	Yes			23.00%
Uganda			79.00%	21.00%	
Coffee, green	14.00%	Yes			11.00%
Tea	38.00%	Yes			29.00%
Tobacco, unmanufactured	50.00%	Yes			39.00%
United Republic of Tanzania			100.00%	0.00%	
Coffee, green	No increase	Yes			No LUC
Tea	14.00%	Yes			14.00%
Tobacco, unmanufactured	52.00%	Yes			52.00%

Table A.8.3 – Overview of land use change calculation for coffee, tea and tobacco for all East African countries. No LUC: no land use change was allocated to the crop.

function of age and 12 additional predictors (geographic distance between sampling sites, disturbance intensity, elevation, biome, latitude, SG–OG patch connectivity, SG and OG patch size, restoration method, realm, differences in sampling effort, number of sites per treatment and taxon). The study employed Generalized Linear Models (GLMs) and multi-model averaging to derive partial effects of each predictor. We spatially mapped recovery time predictions for Sorenson similarity based on weighted parameters of the "general" model (i.e. the simplest model including all data, but omitting the effects of patch size, connectivity and distance). To do this, we first developed spatial datasets for the relevant spatial predictors (Table A.8.4). For non-spatial variables, we assumed partial effects for mammals (taxon), passive restoration (restoration method), equal sampling effort across habitats (sampling effort) and 10 sampling sites per treatment (number of sites). We also added additional partial effects for OG habitat patch size and isolation distance (see Table A.8.4 for details) using the parameters of the "patch" and "distance" models of Curran *et al.* (2014).

Recovery times were defined as the time required for assemblage similarity to fall within half a standard deviation of the average background OG–OG similarity. This allowed for some variability in identifying recovery success given the high level of background variation observed in reference OG–OG comparisons. However, Sorenson similarity (reflecting the presence of shared species) does not differentiate between the type of species contributing to (dis-)similarity. Variation between OG–OG samples may be due to high rates of species turnover of habitat specialists of conservation concern, whereas variation in SG–OG similarity might reflect turnover between habitat specialists and generalist species. Therefore our success criterion of half a standard deviation was statistically conservative to avoid misrepresenting the recovery process. Re-arranging the linear model formula allowed age at recovery to be predicted according to:

$$x_{age} = 10^{\left(\frac{-0.5*s.d.-(-b+\sum_1^n a_i x_{i,j})}{a_{age}}\right)} \tag{8.15}$$

Where a_i is the coefficient for predictor i, n is the number of predictors, x_i is the value of the predictor in cell j, and b is the intercept of the model. Age is back-transformed from $\log_{10}$ space. The resulting spatial recovery time predictions are illustrated in Figure A.8.2.

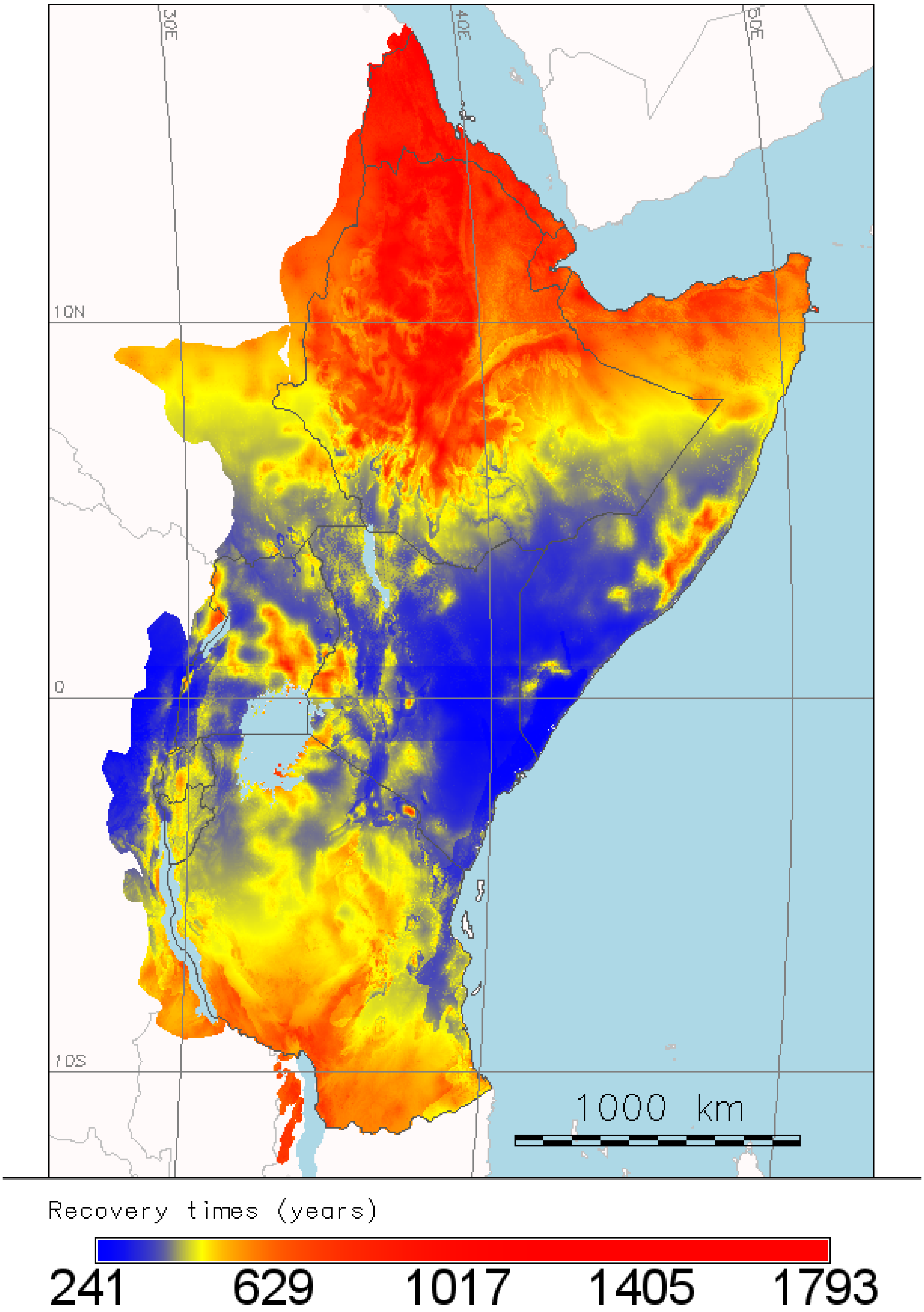

Figure A.8.2 – Predicted recovery time (y) for species similarity the study region. Recovery is assumed to occur when species similarity between secondary and old growth habitats (SG–OG) reaches old growth background (OG–OG) (dis)similarity levels, based on the models of Curran *et al.* (2014).

Predictor	**Preparation**
Elevation	Global 1 km resolution elevation dataset from WorldClim (www.worldclim.org), based resampling the Shuttle Radar Topography Mission (SRTM) 90 m resolution DEM product.
Absolute latitude	Distance from the equator in the North and South direction, in degrees. We assumed no effect of latitude for pixels less that 1 degree from the equator.
Biome	Biome data from the Ecoregion spatial dataset (Olson *et al.* , 2001), with a simple reclassification based on structural complexity: "Open" (biomes 7, 8, 9, 10, 11, 13, 14, 15), "Transition" (biomes 2 and 12), "Broadleaf" (biomes 1 and 4) and "Coniferous" (biomes 3, 5, 6).
Disturbance intensity	We assumed two classes of disturbance intensity for land occupation based on classifying the anthropogenic land cover classes of the ESA GlobCover 2009 land cover product (Bontemps *et al.* , 2011): intensive occupation (classes 10–15 and 190) and extensive occupation (classes 16–32). To be conservative, we assumed land was occupied before regeneration takes place (i.e. no direct regeneration after a once-off transformation, which can lead to faster recovery; Curran et al. accepted)
OG patch size	We added partial effects based on the "patch model" of Curran *et al.* (2014). To do this we assumed all non-anthropogenic land cover classes from the ESA GlobCover 2009 data represents near-natural OG habitat (i.e. source habitat for OG species). We added to this all protected areas for the year 2010 (from the World Database of Protected Areas). We calculated the sum of all natural area in a moving window of 110 pixels dimension (ca. 33 km). We converted these area sums to a $\log_{10}$ scale, and applied partial effects of the model of Curran *et al.* (2014). We did not assume any effect of SG patch size (i.e. the size of the recovering habitat area) as this effect was predicted to be of minor importance in the model of Curran *et al.* (2014) relative to OG patch size.
Distance	The effect of patch isolation on recovery speed was included by calculating the distance from the nearest pixel of (near-)natural habitat (see "OG patch size" description). We applied a partial effect of distance from the "distance" model of Curran et al. (in press), reflecting distance decay in ecological similarity (Soininen *et al.* , 2007). This assumes that more isolated sites will recover more slowly because the similarity between the original community and remaining patches of natural habitat decreases with increasing distance.
SG–OG connectivity	A binary connectivity effect (connected/isolated) was applied based on a cut-off distance of 5 km from the nearest natural habitat patch (i.e. all pixels within 5 km of natural habitat were considered "connected").

Table A.8.4 – Development of spatial data layers for predicting species assemblage recovery times across the study region, based on the models of Curran *et al.* (2014).

Part B: Additional results

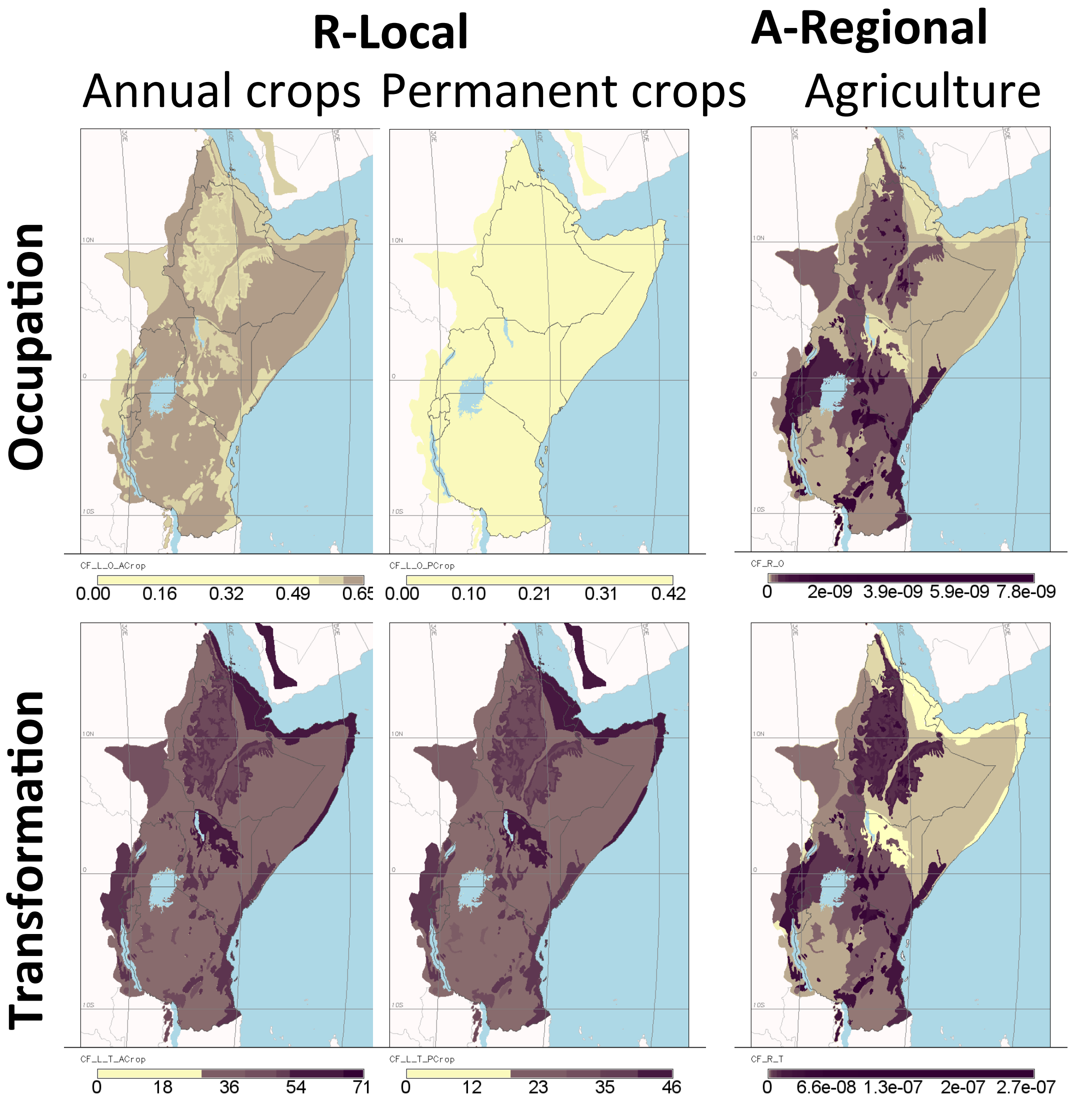

Figure A.8.3 – Biodiversity impacts (characterization factors) for land use (occupation) and land use change (transformation) of the *R-Local* method (de Baan *et al.* , 2013a) and the *A-Regional* method (de Baan *et al.* , 2013b).

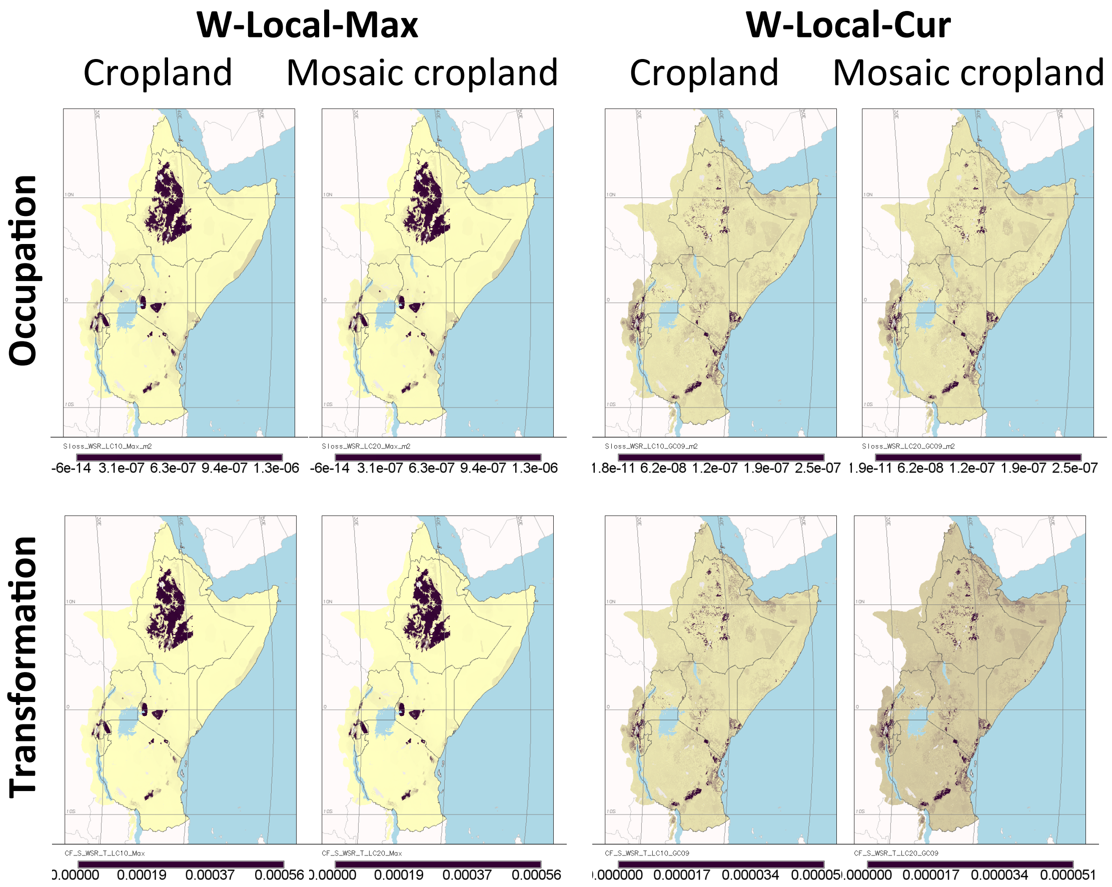

Figure A.8.4 – Biodiversity impacts (characterization factors) for land use (occupation) and land use change (transformation) for the *W-Local* method for the two reference scenarios maximum (*Max*) and current (*Cur*) and the two land cover types.

	Inventory	R-Local	A-Regional	UW-Local-Max	UW-Local-Cur	W-Local-Max
R-Local	0.85 – 1.00					
A-Regional	0.14 – 0.45	0.06 – 0.7				
UW-Local-Max	0.50 – 0.95	0.50 – 0.90	0.28 – 0.51			
UW-Local-Cur	0.28 – 0.72	0.27 – 0.61	0.03 – 0.22	0.27 – 0.73		
W-Local-Max	-0.06 – 0.07	-0.06 – 0.08	0.01 – 0.10	-0.02 – 0.15	-0.07 – -0.01	
W-Local-Cur	0.03 – 0.05	0.03 – 0.06	0.05 – 0.15	0.04 – 0.10	0.06 – 0.12	0.00 – 0.04

Table A.8.5 – Correlation found among the different methods and the inventory data per crop (tea, coffee or tobacco) and per impact type (land use or land use change). The smallest and largest Pearson's correlation coefficients are displayed.

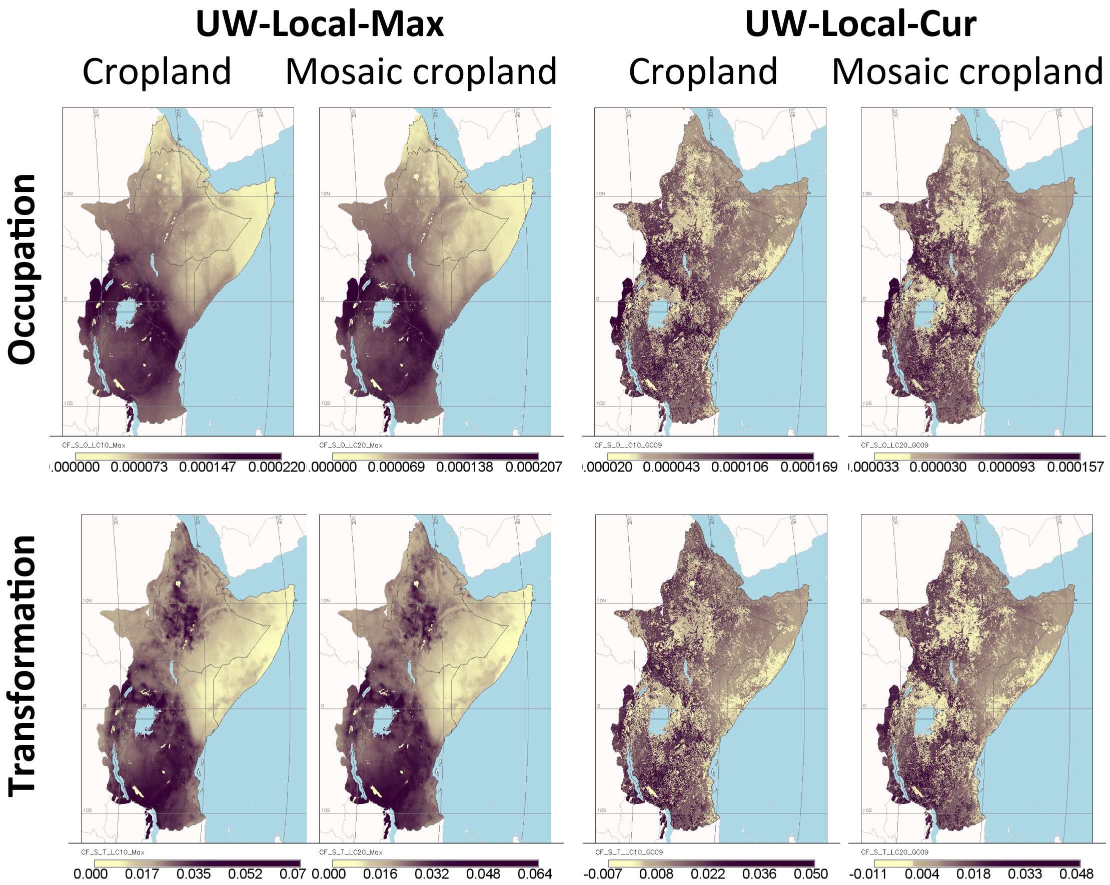

Figure A.8.5 – **Biodiversity impacts (characterization factors) for land use (occupation) and land use change (transformation) for the *UW-Local* method for the two reference scenarios maximum (*Max*) and current (*Cur*) and the two land cover types.**

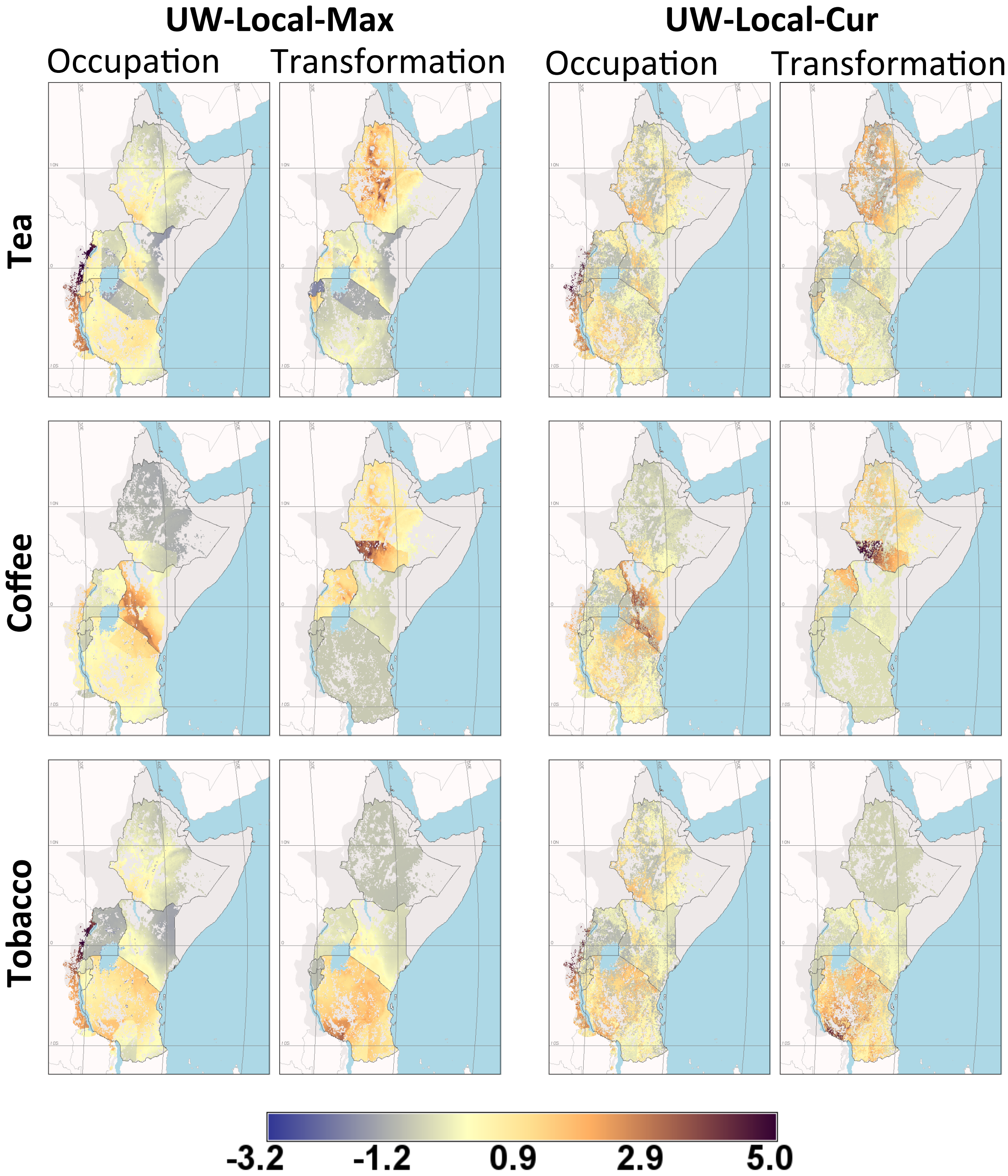

Figure A.8.6 – Final LCA result based on unweighted local species loss (*UW-Local*). Biodiversity loss caused by the land use (occupation) and land use change (transformation) of 1 kg of crop. Assessed with the two reference scenarios maximum (*Max*) and current (*Cur*). Numbers represent deviations from the mean values of each map (0=mean, -1=one standard deviation smaller than the mean, +1= one standard deviation larger than the mean). Values were capped at +5 standard deviation.

8.4.6 References

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8.5 Appendices for Chapter 6

8.5.1 Calculation of yield, price and production values

To allocate impacts at the product level (i.e. per tonne of commodity), and to estimate changes in farm-gate prices caused by compensation payments, we used agricultural production, yield, price and trade data from the FAO for the year 2010 (FAOSTAT, 2013b,a, http://faostat.fao.org/). To estimate average yield, price and production value data to apply to our land use classes, we classified crops according to their growth form and linked these forms to the relevant Globcover land cover code (LC#): (i) intensively produced ground crops (LC#10), (ii) extensively produced crops (LC#20, LC#30, LC#31, LC#32), herbaceous crops (LC#13, LC#15) (iii) shrub or tree crops (LC#12, LC#16). To derive the average value added per tonne of product for each land cover class, we collected yield and farm gate price data for each country based on FAO data (FAOSTAT, 2013b,c). For each country, we took the production-weighted average farm-gate price for the list of crops likely to occur in the aggregate classes described above (i.e. the price per tonne of "commodity equivalent"). First, we derived a production-weighted price for each land use class (combination of crops) in each country:

$$p_{w,k} = p_{c,k}\left(\frac{x_{c,k}}{\bar{x}_k}\right) \tag{8.16}$$

Where the production-weighted price, p_w, for country k reflects the farm gate price (in 2010 dollars), p, of crop c. The national production volume (in tonnes) is x for crop c in country k, and $\bar{x}$ is the average production volume across all crops produced in the country. We also constructed a simlar production-weighted yield for each country:

$$y_{w,k} = y_{c,k}\left(\frac{x_{c,k}}{\bar{x}_k}\right) \tag{8.17}$$

Where the production-weighted yield, $\mathrm{y_w}$, for country k reflects the weighted yield y of crop c. Average value added per tonne of commodity was calculated simply as the product of the weighted average price and yield:

$$v_k = p_{w,k}y_{w,k} \tag{8.18}$$

Where v is the price per tonne of commodity equivalent in country k. For the East African scale this process was conducted for all countries in the region and averaged across countries. For urban areas, we considered the price of urban development land based on a land price model from Central Kenya (Curran *et al.* , n.d.a). The resulting premiums associated to crop-equivalents (and urban development) for each land use class, as well as median impacts and offset ratios for the hypothetical conservation project, are presented in Table A.8.1.

8.5.2 Recovery time predictions and net present biodiversity value

To model restoration and recovery times, we used the models of Curran *et al.* (2014) and the predictors of elevation, latitude, realm, potential biome (using biome classes for WWF Ecoregions of Olson *et al.* , 2001), land use intensity and taxon. For land use intensity, we developed an index based on a reclassification of the ESA GlobCover 2009 land cover product. We created four broad intensity classes: *built areas* (LC#190), *intensively used land* (LC#10–15), *extensively used land* (LC#16–32), and *natural habitat* (remaining LC codes). We assumed restoration could only occur on intensively and extensively used land, excluding built land from the scenarios (i.e. because of irreversible damage). We assumed a regime of active restoration in our scenarios (see Curran *et al.* , 2014 for details). We added corrections (adjusted model coefficients from the "patch dynamics" and "distance" models of Curran *et al.* , 2014) for patch size of natural habitat in a 50 km neighbourhood of each cell (reflecting amounts of "source habitat" for immigration of individuals; Hanski & Gilpin, 1998) and the distance to the nearest natural habitat patch or protected area (to account for isolation effects on dispersal rates resulting in distance decay of ecological similarity; Soininen *et al.* , 2007).

We used the average predicted recovery time across all taxa (birds, mammals, herpetofauna, insects, other invertebrates, plants and trees) to represent the time lag in the delivery of conservation value. Following Overton *et al.* (2013), we applied time discounting to predictions of future conservation value at maturity to derive net present estimates as:

$$p_{rest,i} = WR_{gain,i} * r(1-d)^{t_i} \tag{8.19}$$

Where $p_{rest,i}$ is the present conservation value of cell i for restoration, WR_{gain} is the potential conservation value of the cell, r is the restoration failure rate, d is the discount rate and t_i is the planning horizon, which, under an assumption of habitat restoration equals the predicted recovery time for pixel i. For averted loss, we used a similar formula, but omitted the risk of restoration failure and set t_i to the project time horizon of 100 y.

$$p_{prot,i} = WR_{gain,i} * (1-d)^{t_i} \tag{8.20}$$

Scale	LC #	LC Name – ESA Globcover 2009	Area [ha]	Median Impact	Offset Ratio [O:I]		Farm-gate Price Premium [%]			
				[WR_{marg}/ha]	No-PA	+PA	CONS	PPA	+DEV	ALL
E. Africa -	10	Cultivated And Managed Areas	11,998,611	4.12E-07	7.73	9.64	38.3	43.0	88.3	87.9
Regional scale	12	Post-flooding Or Irrigated Shrub Or Tree Crops	162	3.00E-08	0.56	0.70	3.3	3.7	7.6	7.6
	13	Post-flooding/Irrigated Herb. Crops	567	3.83E-07	7.17	8.94	25.5	28.7	58.8	58.6
	15	Rainfed Herbaceous Crops	43,497	2.96E-07	5.54	6.91	19.7	22.2	45.5	45.2
	16	Rainfed Shrub/Tree Crops	11,664	7.88E-07	14.78	18.43	86.8	97.6	200.3	199.3
	20	Mosaic Crops (50-70%) / Veg. (20-50%)	34,851,141	3.67E-07	6.88	8.58	38.5	43.3	88.9	88.5
	30	Mosaic Veg. (50-70%) / Crops (20-50%)	66,082,392	2.70E-07	5.07	6.32	28.4	31.9	65.5	65.2
	31	Mosaic Grass. Or Shrub. (50-70%) / Crops (20-50%)	972	1.07E-07	2.00	2.50	10.1	11.4	23.3	23.2
	32	Mosaic Forest (50-70%) / Crops (20-50%)	1,617,732	1.19E-06	22.31	27.83	125.0	140.5	288.4	287.0
	190	Artificial Surfaces (Urban > 50%)	273,375	3.67E-07	6.89	8.59	3.3	3.7	7.5	7.5
		Average		**4.21E-07**	**7.9**	**9.8**	**37.9**	**42.6**	**87.4**	**87.0**
Kenya -	10	Cultivated And Managed Areas	1,685,691	6.84E-07	12.82	15.99	28.4	32.0	65.6	65.3
National scale	13	Post-flooding/Irrigated Herb. Crops	486	3.83E-07	7.17	8.94	8.2	9.2	18.9	18.8
	15	Rainfed Herbaceous Crops	16,038	7.96E-07	14.92	18.61	17.0	19.1	39.2	39.0
	16	Rainfed Shrub/Tree Crops	4,617	8.20E-07	15.37	19.17	35.5	39.9	81.9	81.5
	20	Mosaic Crops (50-70%) / Veg. (20-50%)	3,415,608	6.74E-07	12.63	15.76	28.0	31.5	64.7	64.3
	30	Mosaic Veg. (50-70%) / Crops (20-50%)	9,854,541	4.64E-07	8.70	10.85	37.4	42.1	86.3	85.9
	190	Artificial Surfaces (Urban > 50%)	62,208	7.11E-07	13.34	16.63	6.3	7.1	14.6	14.5
		Average		**6.48E-07**	**12.14**	**15.14**	**23.0**	**25.8**	**53.0**	**52.8**
Kenya -	10	Cultivated And Managed Areas	92,016	6.19E-07	11.59	14.46	25.7	28.9	59.3	59.0
Sub-national scale	13	Post-flooding/Irrigated Herb. Crops	486	3.83E-07	7.17	8.94	8.2	9.2	18.9	18.8
	15	Rainfed Herbaceous Crops	5,427	4.03E-07	7.55	9.41	8.6	9.7	19.8	19.7
	16	Rainfed Shrub/Tree Crops	810	8.04E-07	15.07	18.79	34.8	39.1	80.3	79.9
	20	Mosaic Crops (50-70%) / Veg. (20-50%)	427,275	1.03E-06	19.30	24.07	42.8	48.1	98.8	98.3
	30	Mosaic Veg. (50-70%) / Crops (20-50%)	1,977,696	4.53E-07	8.48	10.58	18.8	21.2	43.4	43.2
	190	Artificial Surfaces (Urban > 50%)	35,316	7.11E-07	13.34	16.63	6.3	7.1	14.6	14.5
		Average		**6.29E-07**	**11.79**	**14.70**	**20.7**	**23.3**	**47.9**	**47.6**

Table A.8.1 – Biodiversity impacts per hectare, compensation requirements and farm-gate price premiums of land occupation for 10 ESA Globcover 2009 land use classes at three spatial scales. Farm gate premiums express price increase from compensating one unit of product (one tonne of crop equivalents for agricultural land, 1 ha development land for urban class @ \$20'000 per ha in 2010). Costs cover four scenarios: CONS = only non-PA land considered; PPA = support included for half the management costs of public PA cats 5 & 6; DEV = includes development budget; ALL = includes both development budget and public PA support. Offset ratios based on biodiversity gain data in Table 6.1. O:I = ratio of ha offset to ha impact; NPC = net present costs

Restoration failure risk. To account for the risk of restoration failure, we used modelling results from a recent meta-analysis of habitat restoration (Curran *et al.* , 2014). Curran *et al.* (2014) predict a minimum of 12% of all restoration projects will fail outright based on the proportion of modelling outcomes supporting a *negative* biodiversity-age relationship. This indicates novel and undesired recovery pathways. Although the study also showed much longer recovery times and higher failure risk for recovery of assemblage composition measured with the Morisita-Horn index (reflecting the relative frequencies of species, and not just their presence or absence), we chose the Sorenson model because (i) the model of Curran *et al.* (2014) does not differentiate between restoration methods, which can reduce risks and speed up recovery (Lindenmayer *et al.* , 2012), (ii) targeted restoration would favour areas with lower risks and quicker recovery (e.g. connected to existing old growth habitat) rather than the average trends reported in Curran *et al.* (2014), and (iii) the presence of all old-growth species (Sorenson similarity recovery model) matches the focus on species occupancy (representation) used in the benefit function rather than population abundance.

8.5.3 Habitat loss as basis for the discount rate

We based our discount rate on the rate of habitat loss in Kenya, which has averaged over the past decade 0.32% and 0.26% for forest and non-forest habitat, respectively (FAOSTAT, 2013a). This considers all habitat (i.e. protected and non-protected), does not differentiate between native and planted forest, and does not account for habitat degradation. Kenya has a forest area covering 6.2% of the land surface (3.5 million ha). Only 1.96% (1.106 million ha) is native, and 1.23% (ca. 700'000 ha) is still in an old growth state. Since managed forests are generally replanted, one should expect net loss to be concentrated in native forests. Therefore the effective loss rate for native forest is ca. $0.32\% * \left(\frac{6.2}{1.96}\right) = 1.02\%$. Additional impacts from forest degradation likely exacerbate the effect of this loss rate on biodiversity. At the same time, non-forest habitat appears to be declining at a slower rate. We therefore chose an effective discount rate for averted averted loss offsets of 1%. Our discount rate might be low in consideration of other proxies of biodiversity threat in Kenya. Kenya's average population growth rate over the past 10 y was 2.7%. This implies pressure on land resources will grow to meet the needs of this growing population, and therefore our estimates of current loss rates might be insufficient. Per-capita consumption will also likely increase in the future, which can be approximated by growth in real GDP (expected to be maintained at the current average of 4–5%).

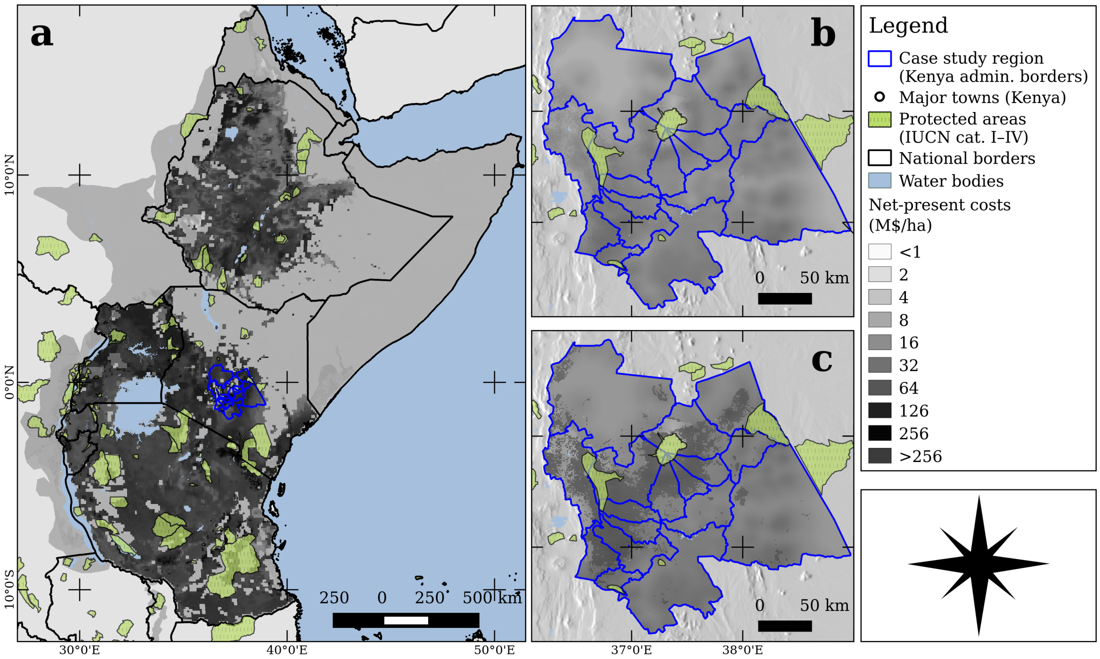

Figure A.8.1 – Net present conservation costs at the regional scale (a) and local scale, excluding (b) and including (c) a development budget. Economic costs reflect discounted investment and operational costs over a 100 y time horizon at a 1% discount rate. Management cost data was estimated only for Kenya and applied across the study region (see Curran *et al.* , n.d.a). For the regional data, an opportunity cost approach was assumed, using gross agricultural returns data from the FAO Global Agroecological Zones, vers. 3. At the local level, a more detailed cost model based on land price data was used (Curran *et al.* , n.d.a). Due to the difference in methods, cost data at the regional and local scale should not be expected to match. The regional data were only used for targeting purposes (i.e. based on relative cost differences between locations), whereas the detailed local models were used for estimating absolute conservation costs of the hypothetical compensation project.

8.5.4 Regional targeting using *Zonation*

We used the additive benefit function under a target-based planning approach to conduct our regional prioritization (step 5, see Section 6.2.5). The ABF is similar to our weighted richness benefit function, WR, but is calculated iteratively at each cell removal step in the *Zonation* process, and only considers the unprotected portion of a species' range in the rarity weight:

$$ABF_i = \frac{1}{c_i} \sum_j w_j q_{i,j} \qquad (8.21)$$

Where c_i is the cost of retaining cell i in the reserve network (i.e. the conservation costs), $q_{i,j}$ is the proportional range equivalent of species j in cell i and w_j is a species weight. The target-based planning approach modifies uses the weight to regulate how species reach their conservation targets (e.g. 25% of the regional range of each species should be protected). As cell removal approaches a species protection target (e.g. 27% of a species' suitable habitat remains in the landscape), the species receives a higher weight, w_j, and thus cell removal shifts to other areas and targets other, better-represented, species.

We used a cell resolution of 1'800 m at the regional scale (due to computational limitations). We considered habitat of both medium and high quality in the mammal suitability models, converting these to a probability of occurrence of 0.5 and 1, respectively. We accounted for existing conservation coverage using a mask of the World Database on Protected Areas (WDPA), 2010 release (UNEP-WCMC, 2010), and excluded built/urban cells from the analysis. As a separate set of ecosystem-level biodiversity features, we included all WWF Ecoregions in the study area, stipulating a target coverage of 25%, identical to targets used at the species level. We also included a habitat condition layer for the Ecoregion data, by modifying the land use intensity map described above. To do this we attributed broad "community intactness" scores to the three aggregated land use classes (built, intensive, extensive) based on a recent review of disturbance effects on biodiversity (Alkemade *et al.* , 2009). We assigned relative scores of 0.2, 0.3 and 0.5 intactness (relative to a natural baseline) for built, intensive and extensive land, respectively (based on the Mean Species Abundance indicator of Alkemade *et al.* , 2009). Finally, we included a layer representing distance to nearest near-natural or protected area (PA) with a negative weight to favour agglomeration with existing reserve networks.

Conservation management costs. Conservation management cost data were gathered by Curran *et al.* (n.d.a) from interviews with four conservation organizations in Kenya, and aggregated for montane forest, montane moorland/grassland and lower rangeland ecosystems. We assumed these costs represented acceptable estimates to extrapolate across the East African subregion, but stress country/locality-specific management cost data would be desirable for future analyses. We did not correct estimates across countries for purchasing power parity. Curran *et al.* (n.d.a) differentiate between investment and operational phases in the establishment of conservation projects, which were combined by discounting future cash flows to current equivalents over the 100 y planning horizon, at a discount rate of 1% (see Curran *et al.* , n.d.a for a discussion on the choice of discount rate). This resulted in net present costs (NPC) of conservation investments according to:

$$NPC = C_{inv} + \sum_t C_{opp}(1-d)^t \tag{8.22}$$

Where C_{inv} and C_{opp} represent investment and operational costs, respectively, d is the discount rate and t is the discreet time-step. The resulting cost estimates are shown in the Figure A.8.1.

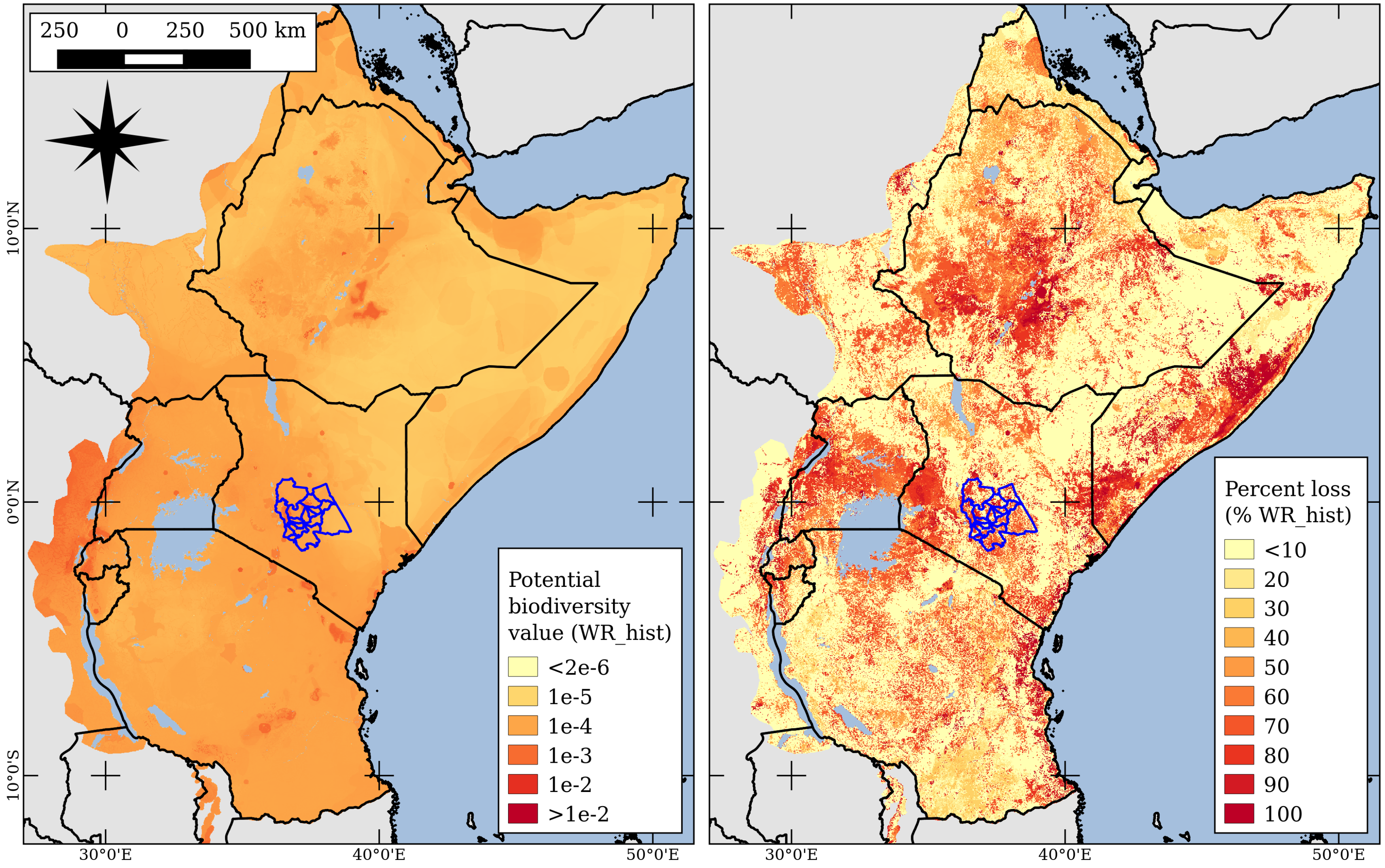

Figure A.8.2 – Potential pattern (left) and percentage reduction (right) in weighted species richness using the WR_{hist} benefit function (rage size weighted richness). The potential pattern represents reference WR_{hist} values in the absence of anthropogenic land use. It was calculated using mammal habitat suitability models from the "maximum species range reference scenario" of de Baan *et al.* (n.d.). Percentage loss shows difference between potential and current states, where current states refers to WR_{hist} values calculated based on contemporary habitat suitability models using data from Rondinini *et al.* (2011a).

8.5.5 Additional results

Alternative species weights (WR_{hist}). To test the sensitivity of our results to choce of benefit function, we also repeated the analysis using another variant of weighted richness, WR_{hist}. In this variant, the weights reflect *historic* changes in conservation value (WR_{hist}), rather than marginal changes (WR_{marg}), as in the main text. The weight for range size was calculated separately for both potential and current biodiversity models (i.e. the potential model used range sizes from the *potential* HSMs, and the current model from the current HSMs), and no threat weight was applied. This a species that has seen a large reduction in its potential range should a much higher current value than potential value. Because of this, WR_{hist} exhibits negative change (i.e. biodiversity gain) between potential and current state. This is due to the increase in marginal value of habitat harbouring species that have lost much of their former range. For our impact assessment, we treated negative values as zero values to remove this artifact of the method.

Results for WR_{hist} are shown for both biodiversity impacts and price increases (Figs. A.8.3 and A.8.4), along with results at different spatial scales (Figs. A.8.5 and A.8.6). In general, premiums were predicted to be lower than with WR_{marg}, with median values ranging up to *ca.* 15% and 35% for the cheapest and most expensive scenarios, respectively, and upper quartile reaching 25% and 60%, respectively (Appendix 8.5.5, Fig. A.8.4).

Log-transformed mean data ($\mathbf{WR_{marg}}$). To provide an alternative to using median data, we accounted for skew in our impact and gain data by log-transforming values and recalculating results (see Fig. A.8.7). With log-normal mean data for impacts and gains, price premiums predicted to be lower than median values for both methods (by about a factor of 5 for WR_{marg} and 2 for WR_{hist}) leading to upper premium ranges of *ca.* 60% (Fig. A.8.7).

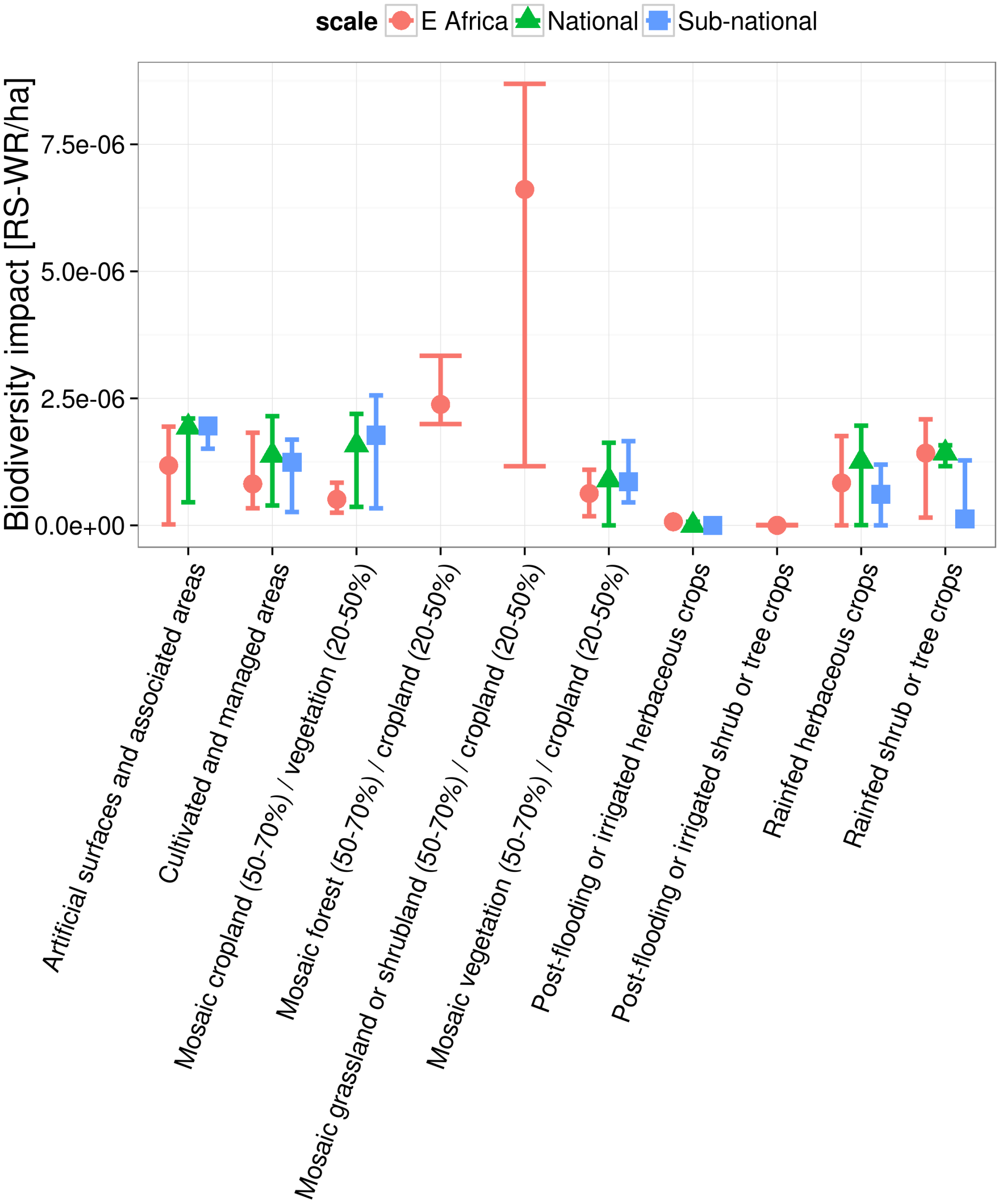

Figure A.8.3 – Biodiversity impacts of ten ESA Globcover land use classes at three spatial scales using the WR_{hist} method. Plot shows median impacts and inner quartiles (25 and 75 percentiles) per ha for each land use class. Missing data for specific scales (e.g. “Mosaic grassland...” at National and Sub-national scale) indicate no presence of land use class at that scale. RS-WR = range size-weighted richness (WR_{hist}).

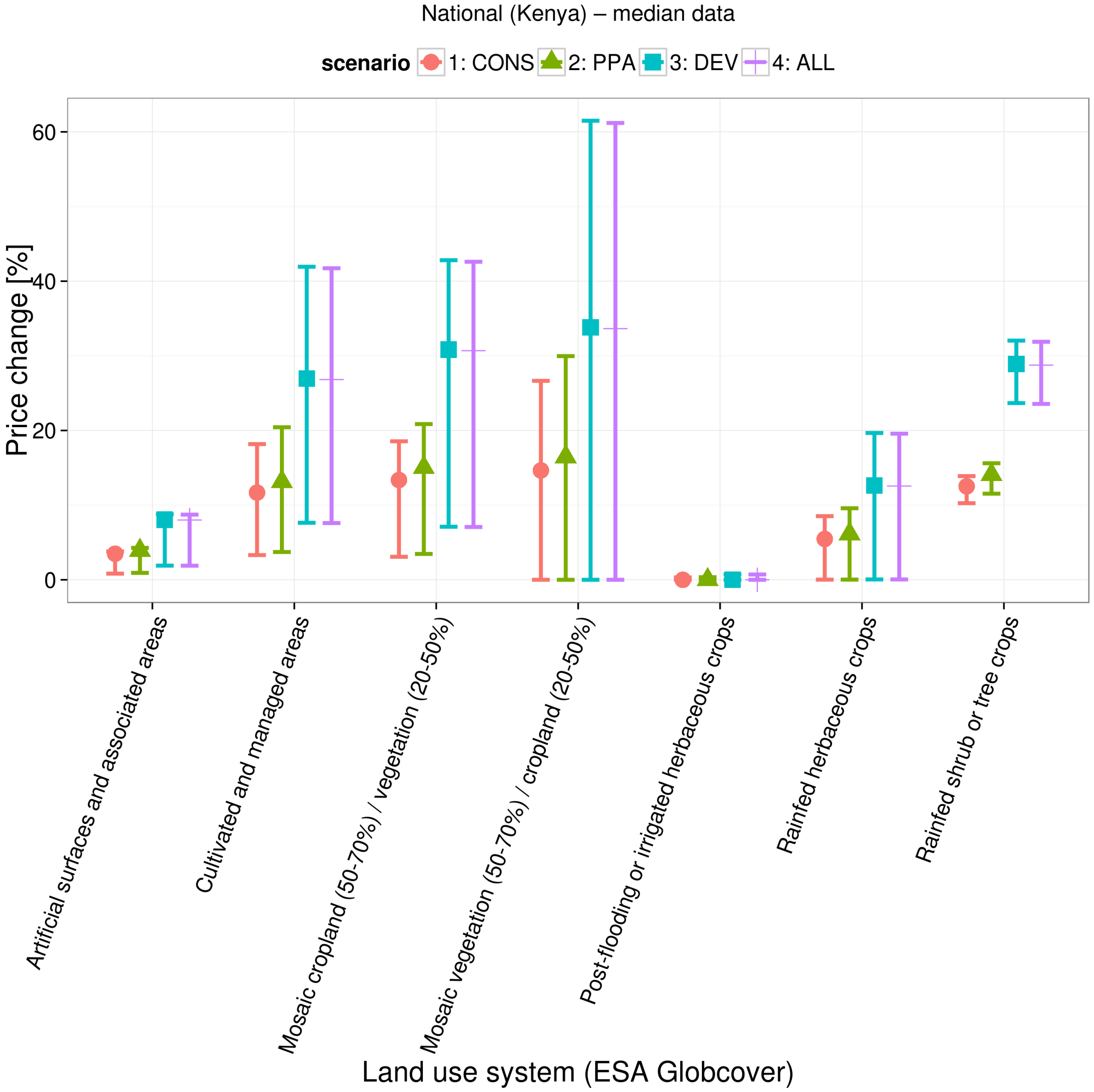

Figure A.8.4 – National scale predicted farm-gate price premiums for land use products (tonne of crop equivalent, ha of development land) for ESA Globcover land cover classes using the WR_{hist} method. Results shown for four cost scenarios: CONS = only non-PA land considered; PPA = support included for half the management costs of public PA cats 5 & 6; DEV = includes development budget; ALL = includes both development budget and public PA support. Data range represents median and inner quartiles (25 and 75 percentiles).

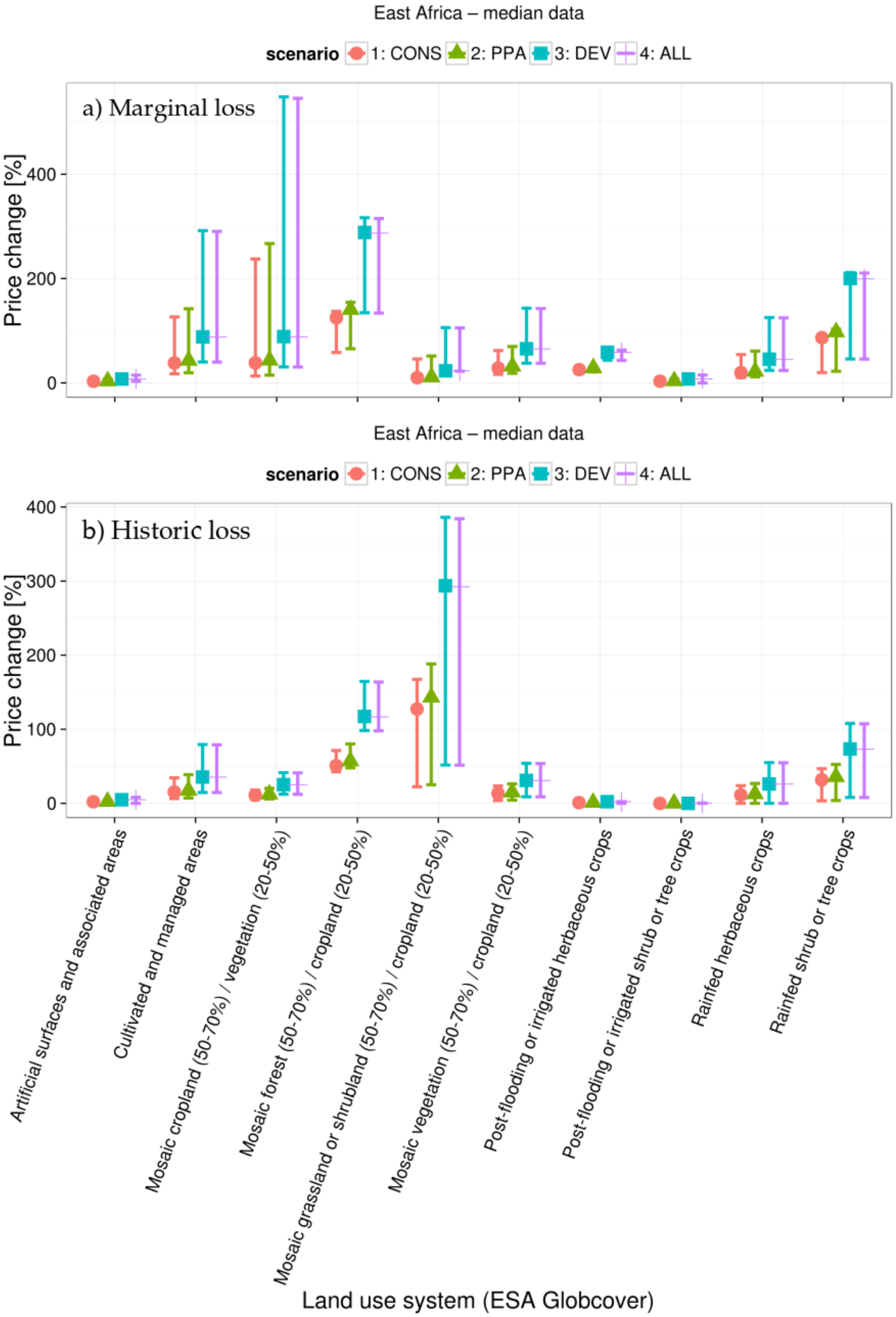

Figure A.8.5 – Regional (East Africa) scale predicted farm-gate price premiums for land use products (tonne of crop equivalent, ha of development land) for ESA Globcover land cover classes using both WR_{marg} (a) and WR_{hist} (b) methods. Results shown for four cost scenarios: CONS = only non-PA land considered; PPA = support included for half the management costs of public PA cats 5 & 6; DEV = includes development budget; ALL = includes both development budget and public PA support. Data range represents median and inner quartiles (25 and 75 percentiles).

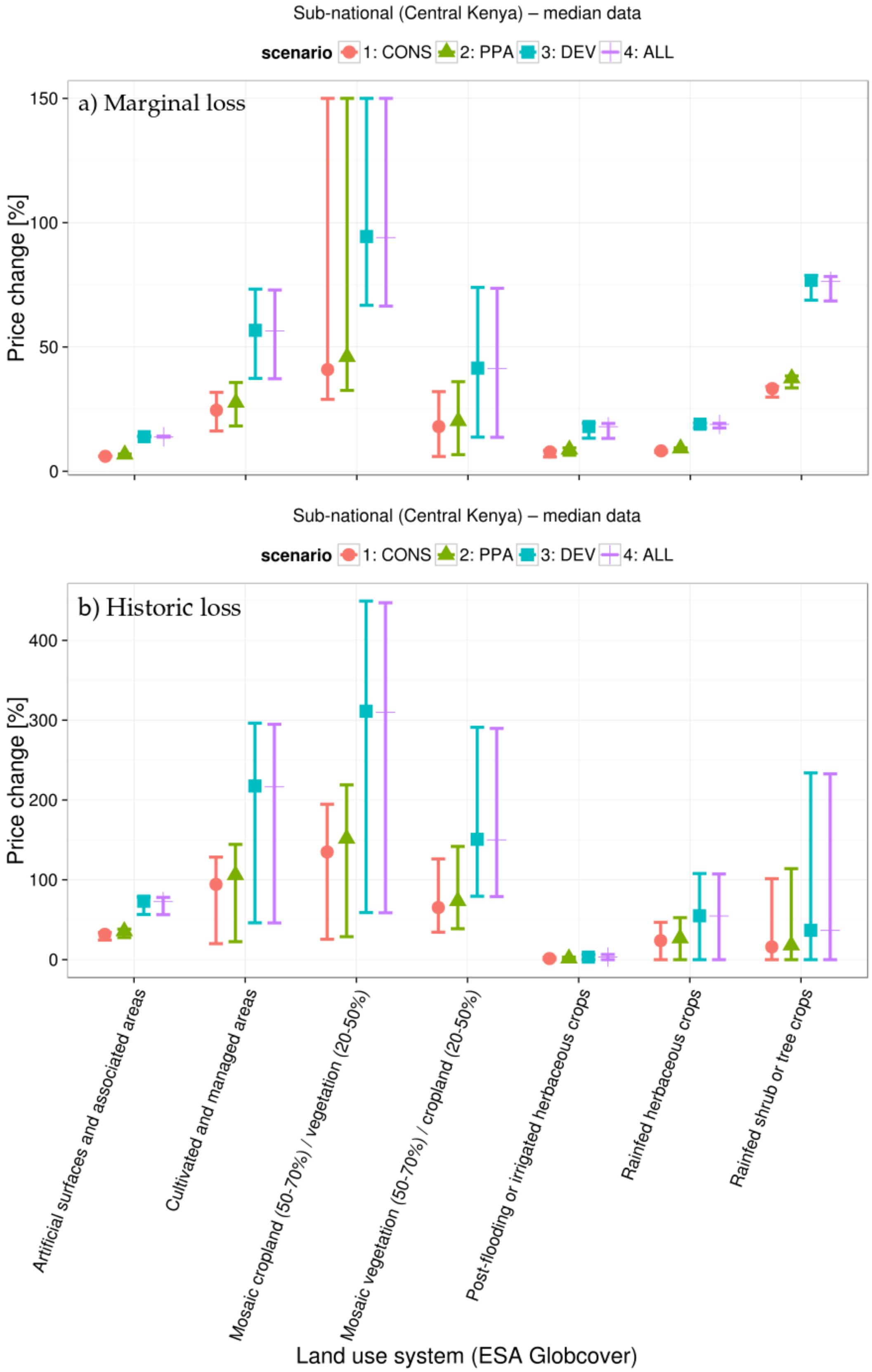

Figure A.8.6 – Sub-national scale predicted farm-gate price premiums for land use products (tonne of crop equivalent, ha of development land) for ESA Globcover land cover classes using both marginal (WR_{marg}: *a*) and historic (WR_{hist}; b) methods. Results shown for four scenarios: CONS = only non-PA land considered; PPA = support included for half the management costs of public PA cats 5 & 6; DEV = includes development budget; ALL = includes both development budget and public PA support. Data range represents median and inner quartiles (25 and 75 percentiles).

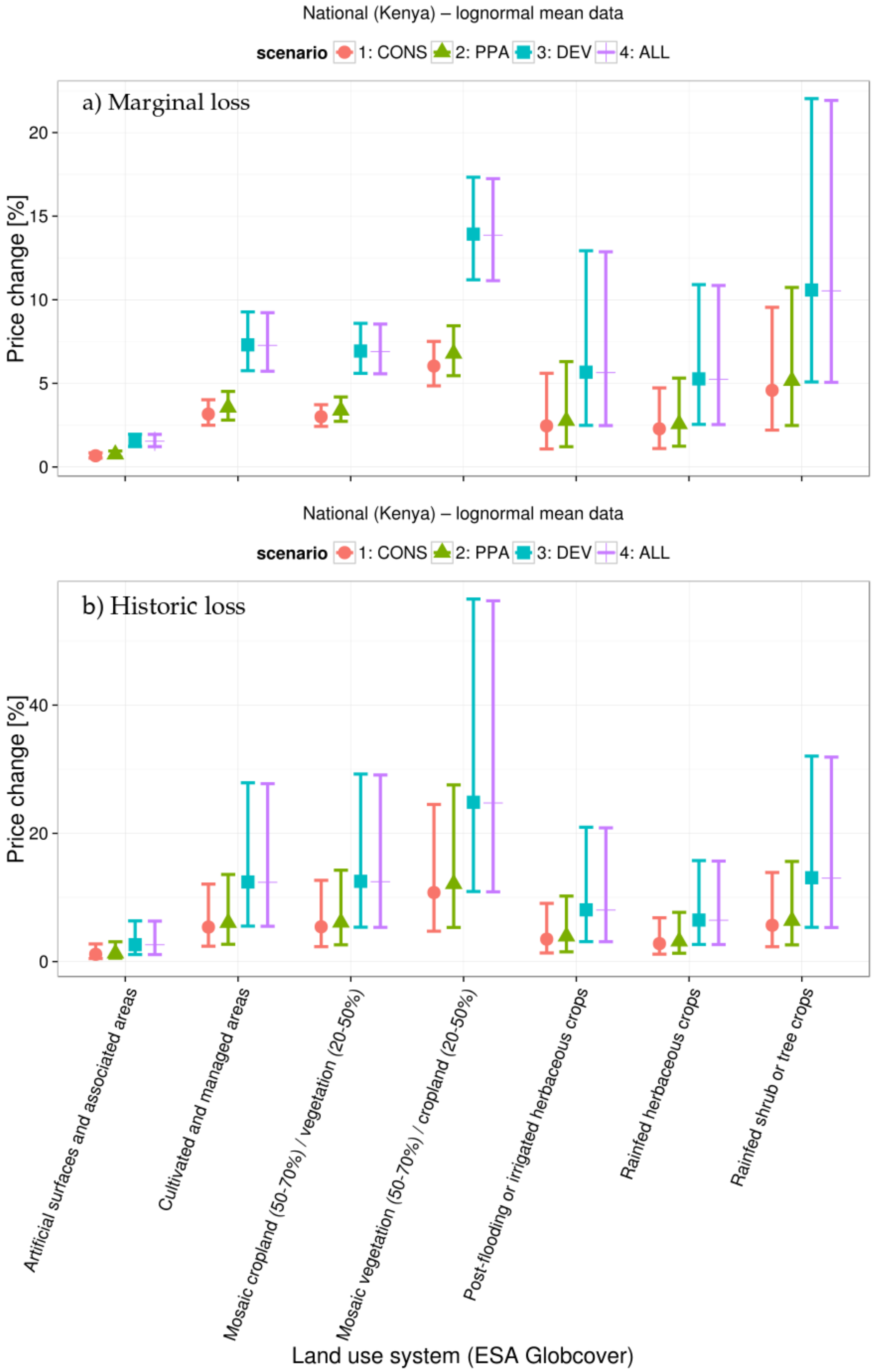

Figure A.8.7 – **National scale predicted farm-gate price premiums using log-transformed mean impact/gain data.** Values expressed per land use product (tonne of crop equivalent, ha of development land) for ESA Globcover land cover classes using the marginal (a) and historic (b) loss methods. Results shown for four cost scenarios: CONS = only non-PA land considered; PPA = support included for half the management costs of public PA cats 5 & 6; DEV = includes development budget; ALL = includes both development budget and public PA support. Data range represents median and inner quartiles (25 and 75 percentiles).

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