STRANDED ASSETS

PROGRAMME

SMITH SCHOOL OF ENTERPRISE AND THE ENVIRONMENT

UNIVERSITY OF OXFORD

A Framework for Protected Area Asset Management

December 2015

WWF

Luc Hoffmann Institute

UNEP WCMC

ZSL

Principal Contributors Paul Jepson | Ben Caldecott | Harriet Milligan | Dexiang Chen

About the Project for Protected Area Resilience

The Project for Protected Area Resilience (PPAR) seeks to reinvigorate the Protected Area (PA) conservation discourse. It examines the monetisable and non-monetisable values PA assets generate, who creates or captures this value, and how more value can be generated through new public, private, and philanthropic investments. The project is also concerned with how to safeguard PAs in light of current and emerging risks threatening their ability to generate value sustainably – in other words we want to avoid PA assets becoming 'stranded assets'. In addition, the project is looking at how to prioritise different types of PA funding and how PAs can maximise their impact given limited public funds. To find out more, visit: http://www.smithschool.ox.ac.uk/research-programmes/protected-area/index.php

The project is led by Ben Caldecott and Paul Jepson at the University of Oxford's Smith School of Enterprise and the Environment. PPAR's advisory board currently includes: Justin Adams (Managing Director, Global Lands, TNC), André Abadie (Managing Director/Head of Global Environmental & Social Risk Management, J.P. Morgan), Professor Jonathan Baillie (Conservation Programmes Director, ZSL), Robin Bidwell (The Woodchester Trust), Glyn Davies (Director of Programmes, WWF-UK), Christian del Valle (Managing Partner, Althelia Ecosphere), Rupert Edwards (Senior Adviser, Forest Trends), Professor Marc Hockings (Head of Science, World Commission on Protected Areas, Program Director of Environmental Management, University of Queensland), Naomi Kingston (Head of Protected Area Programme, UNEP-WCMC), Kathy MacKinnon (World Commission on Protected Areas), Stephanie Maier (Head of Corporate Responsibility, Aviva Investors), Therese Niklasson (Head of ESG Research, Investec Asset Management), Sue Stolton (Director, Equilibrium Research), Joshua Tewksbury (former Director, Luc Hoffmann Institute), Francis Vorhies (Director, Earthmind), and Sir Graham Wynne (Special Adviser, The Prince of Wales' International Sustainability Unit).

The project is currently supported by grants from WWF-UK, The Luc Hoffmann Institute, and The Woodchester Trust. UNEP-WCMC and ZSL are collaborating research partners.

About the Principal Contributors

Paul Jepson directs the University of Oxford's MSc in Biodiversity, Conservation, and Management and leads an interdisciplinary Conservation Governance Lab. He has a doctorate in protected area policy and previously worked for government and NGOs on protected area planning and management in the UK, Indonesia, and Indochina.

Ben Caldecott is a Programme Director at the University of Oxford's Smith School of Enterprise and the Environment, where he founded and directs the Stranded Assets Programme. He is concurrently an Adviser to The Prince of Wales's Accounting for Sustainability Project and an Academic Visitor at the Bank of England.

Harriet Milligan is a postdoctoral researcher at the Institute of Zoology, ZSL, Harriet works on understanding population trends of vertebrate populations within protected areas globally. This involves the use of a large dataset of time series data and a generalised additive modelling framework to create the Living Planet Index (LPI). The LPI has been adopted by the Convention of Biological Diversity as an indicator of progress towards 2011-2020 targets. Current research is based on using the LPI to compare abundance change in over time for 'protected' versus 'not protected' populations of the same species at the country level.

Dexiang Chen is a Research Assistant at the University of Oxford's Smith School of Enterprise and the Environment. He has a Master's degree in Biodiversity, Conservation, and Management from the University of Oxford. Dexiang is interested in the nexus between biodiversity and business. Dexiang has held previous roles in government, civil society, and academe with experience in reef restoration, environmental impact assessments and systematic conservation planning projects in Southeast Asia.

Acknowledgements

The authors would especially like to thank the project's advisory board and collaborating research partners. We would also like to give special recognition to the supporters of the project, in particular WWF-UK, The Luc Hoffmann Institute, The Woodchester Trust, and the Brazilian National Council for Scientific and Technological Development.

We would also like to thank Susanne Schmitt, Richard Ladle, Ricardo Correia, Peter Long, Gordon Clark, Juliana Stropp, Marcello Gonclaves, Marcello Robin Freeman, Noelle Kumpul, Michael T'Sas-Rolfes, Ariel Gregory, and Natalie Page for valuable discussions and inputs that have supported the development of the asset framework.

Table of Contents

SMITH SCHOOL OF ENTERPRISE AND THE ENVIRONMENT

UNIVERSITY OF OXFORD

Table of Contents Continued

Executive Summary

- This report sets out a new asset framework for protected areas (PAs), that involves five typologies, namely for investment, situated assets, forms of value, value capture, and risk factors.

- At the 2003 World Parks Congress, WWF CEO Claude Martin commented that PAs represent, 'the largest conscious land use change in history'. The total land area with some form of protection tripled between 1985 and 2000 and is now 12.5% (Watson et al, 2014). A key focus of the 2014 World Parks Congress was on how to meet Aichi Target 11 – that by 2020 the area should increase to 17% (and 10% of marine areas) and relatedly, how to ensure appropriate governance and effective management of this expanded global PA estate.

- The resilience of PAs will increasingly depend on their perceived contribution to society and economy. In an era of growing population, struggling economies, increasing resource extraction and expanding linear infrastructure development there is a risk that they could be seen as being 'in the way' of human development. Indeed some countries are backtracking on international PA commitments, and PA downgrading, downsizing, and degazettement (PADDD) has recently emerged as a topic of concern (Mascia and Pailler, 2011). Budgets for PA management are also being cut (Watson et al, 2014). In short, PAs are facing increased political and social vulnerability.

- The future of PAs as a cultural and policy ideal cannot be taken for granted. In order to assure the future of PAs in increasingly risky and volatile contexts three things need to happen simultaneously: i) we need to demonstrate the value generated by PAs in ways that are meaningful for citizens, politicians, and markets in a rapidly changing world; ii) we need to understand better the forms of value generated by PAs to enable enhanced risk management; and iii) we need to attract new investment into PAs from old and new funding sources.

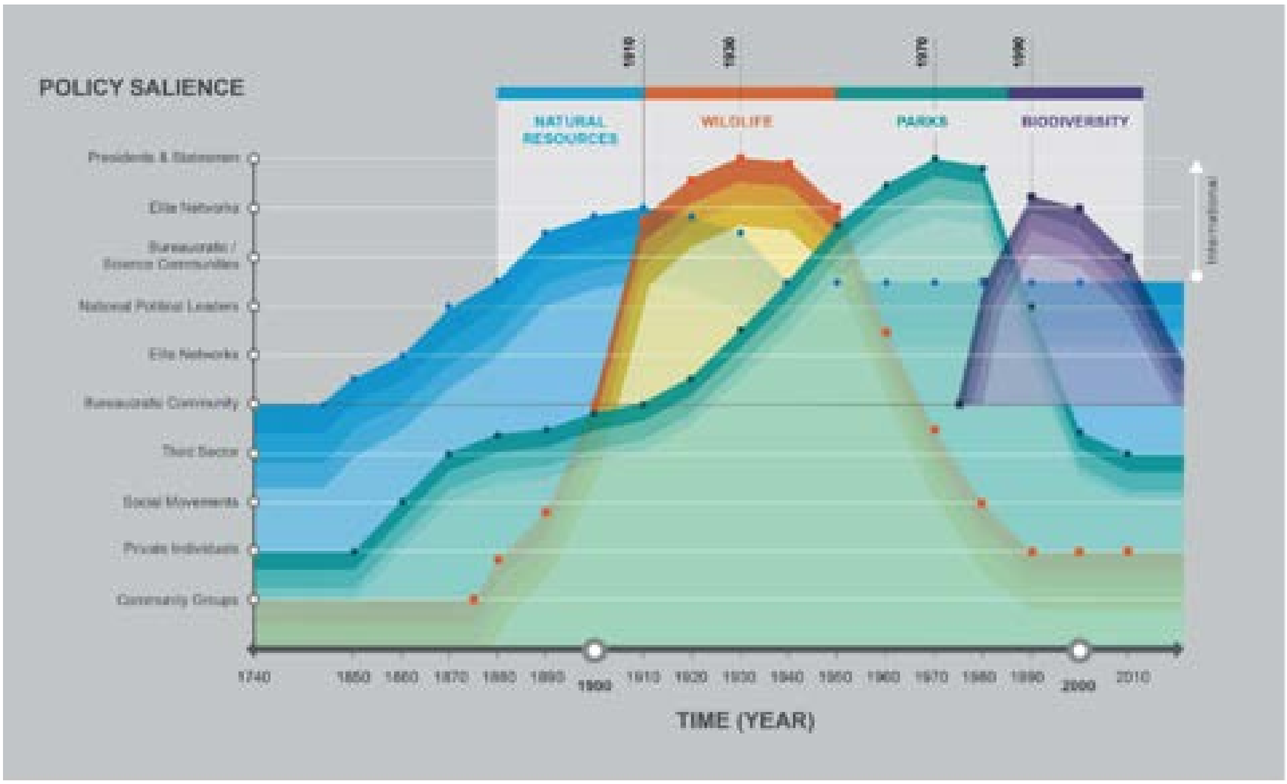
- Framing PAs as a spatial asset is a means to restate the case for protected areas. We believe that the social and political resilience of PAs will be increased if the different forms of value they generate, and for whom, are specified along with the forms of investment needed to generate value in the medium and long term. Our new asset framework represents a heuristic tool for identifying the diverse forms of value that accrue to nations, society, people, and economies from PA assets (many of which cannot be monetised), where value resides and the relationship to investment in a range of asset types. In so doing, our framework identifies value that is being captured and potential value that has yet to realised, thereby creating a case for new investment in PAs.

- An asset-based approach to protected areas would enable decision-makers and investors to optimise their PA assets – as portfolios or networks – in terms of their value generating purpose, spatial locations and investment profile. Our asset framework could be deployed to translate PA management plans into PA business and investment plans and more broadly to identify and consolidate links between PAs, heritage conservation, ecosystem restoration, and other areas or public policy – notably health, recreation, water, and green development. Furthermore, our framework is already guiding and inspiring work on the development of a new generation of PA metrics afforded by 'big data' and new computational techniques. Such metrics have the potential to support improved decision-making in protected area asset management – that effectively values, maintains, operates, and defends PA assets; attracts and properly deploys the right mix of private, public, and philanthropic capital into PA assets; and also identifies assets that are under-performing or poorly utilised.

Part I: A framework for Protected Area asset management

The creation of protected areas (PAs) for conserving nature over the long term is one of the defining features of the 20th century: the total terrestrial land area with some form of protection rose from <2% in 1900 to 12.5% as of now (Watson et al, 2014). PAs in their various forms have shaped culture and society and are a cornerstone of efforts to sustain the Earth's biodiversity and ecosystems.

Figure 1: *Waves of protected area policy*



PAs are at increasing risk on a number of fronts. A combination of population growth, competing claims for land resources, growing demands for natural resources, and the expansion of infrastructure has resulted in diminished political, policy, and public support for PAs. Increasingly PAs are seen as a luxury that struggling economies can ill afford and/or are in conflict with other policy priorities, such as economic development. This is at a time when many PAs are facing enhanced risks from climate change, invasive species, and managers are struggling to be effective in the face of significant funding shortfalls – estimated at between US$1 billion and US$1.7 billion per year in developing countries (Bruner et al, 2004).

The impetus for establishing PAs is centuries old. Motivations vary over time and place but all have been concerned with protecting and developing biophysical assets to generate forms of individual and societal value. PA histories identify four waves of PA policy – periods when PAs assumed policy salience across multiple domains of society (see Figure 1). The policy frameworks associated with each wave offer a rich source of insight for restating the case for PAs in 21st-century terms. They also remind us that the imperatives of international policy are mostly utilitarian – natural resources, natural disaster mitigation, ecosystem services, livelihoods, and jobs. As a result, contemporary PA discourse, metrics, and models reflect this.

But there is a significant weakness – international conservation policy does not 'do' national beautification, outdoor recreation, and wildlife conservation, or at least, not in any meaningful way. Yet these are the frameworks that are most rooted in society and that are manifest in campaigns and other forms of reputational risk to politicians, businesses, and markets. Our understanding of PAs and our approach to them must reflect these concerns too, and at present they do so inadequately.

In some parts of the world there have been moves to downgrade, downsize, or degazette protected areas (PADDD). The conservation community is struggling to know how to respond. For example, the legitimate request from the extractive sector and investment markets for clear guidance on which PAs should be 'no-go' exposes two difficult questions – who should decide and on what basis?

Political and economic pressures on PAs are exacerbating old tensions within conservation policy, between those who believe we need PAs to be flexible and accommodate new imperatives and those who argue that PAs have been hard fought for and should be inviolate – that if we give ground we could lose momentum and never regain the initiative. Such internal tensions compound the risk: we need a framework for PAs that generates unity of vision and purpose within conservation and with other sectors that are acting to shape the future of the planet.

In order to assure the future of PAs in increasingly risky and volatile contexts three things need to happen simultaneously:

i) we need to demonstrate the value generated by PAs in ways that are meaningful for citizens, politicians, and markets in a rapidly changing world;
ii) we need to better understand the forms of value generated by PAs to enable enhanced risk management; and
iii) we need to attract new investment into PAs from old and new funding sources.

This report substantiates a new approach that can help to deliver the above outcomes. Part I sets out concisely the core elements of our new framework and its relevancy and suitability. We also apply our framework to two country case studies: Brazil and Tanzania. Part II contains more of the underlying research, literature, and data that underpins Part I. This has been separated out to ensure that the main conclusions in Part I could be presented in a concise manner. Part II also presents more detail on the approaches we used and literature we systematically reviewed. Here we also offer a richer snapshot of the values that PAs generate, the investments being made into PAs, and the current and emerging threats they face.

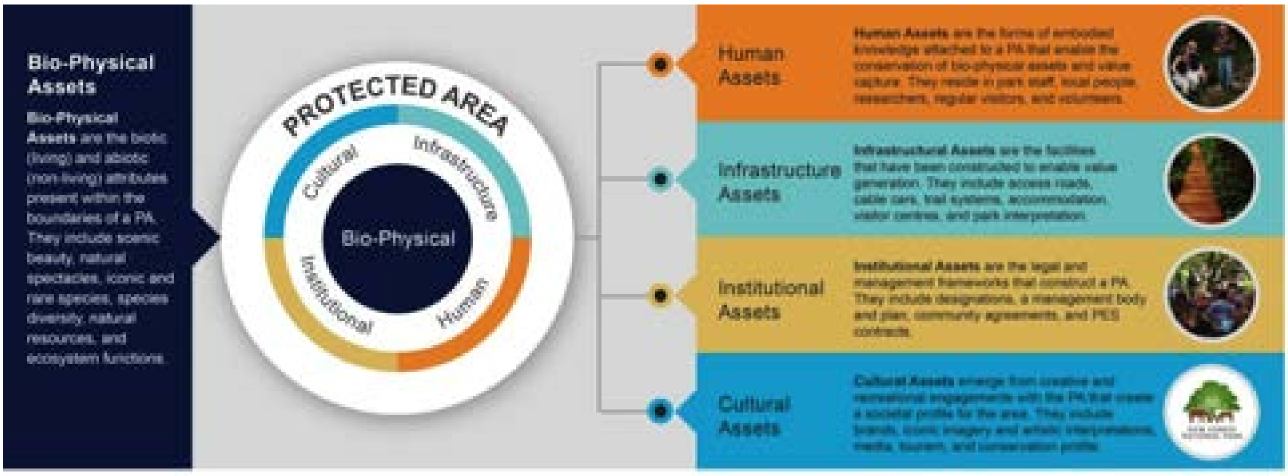
Section 1: Protected Areas as an asset class

Authors: Paul Jepson, Ben Caldecott, and Richard J. Ladle

Our asset framework captures and formalises the types of value generated by investments in PAs in different places over time. PAs are a spatial asset class created through investments in a range of asset types (see Figure 1.1 below). The 'package' of asset types embodied in a PA will differ between bioregion, country, and era of establishment.

The interaction of different asset types linked to a PA generates different combinations of value that accrue to, or can be captured by, different groups in society. Crucially key asset types and the forms of value they produce are created jointly by the people engaging with PAs. For example the Slovenian Alpine club invested in trail infrastructure and maintaining cultural traditions during the Soviet era. Following independence from Yugoslavia, Mont Triglav became a powerful national icon and climbing the mountain a symbolic act expressing Slovenian identity. Value in the form of unity and identity is immense and recognisable, but non-monetisable. Another famous example is the individual investments by early landscape photographers in capturing evocative, artistic images of the Yosemite valley, California. These cultural assets interacted with other forces to generate diverse forms of value over time: wilderness as part of an American cultural identity distinct from Europe, outdoor recreation, and tourism economies. Apple have named a version of their operating system 'Yosemite' illustrating that forms of value accrued from PAs are free cultural assets that can be captured for purposes unimaginable at the time of initial investment.

Figure 1.1: *Protected Area Assets*



Since the early 1990s the dominant policy framework for PAs has been biodiversity conservation (Haila & Kouki, 1994). For many this is an end in itself: yet after more than 25 years the forms of value that biodiversity assets generate for people, economy, and ecosystems have yet to be articulated in a sufficiently compelling way. In our asset framework biodiversity is one type of biophysical asset. By foregrounding other biophysical assets our framework represents a more holistic and comprehensive summary of the forms of value captured by four key interlocking domains of society: i) nations and polity – the process and conventions through which society is organised; ii) economy and enterprise – the production and supply of goods and services that generate money and jobs, iii) organisations – the groups with a particular purpose that do things, and, perhaps most importantly, iv) citizens – inhabitants of a nation who aspire to quality lives.

The list of values that each of these domains captures, or could capture, from PAs is large, diverse, and significant and accrues over long timescales. Importantly, our proposed framework restates the value PAs generate for urban citizens and for nations and regions. These were explicit in the 'wildlife' and 'parks' waves of protected area policy (see Figure 1.1), but are overshadowed in contemporary biodiversity and ecosystem services discourse.

Our framework strengthens accountability in the international protected area regime. This is because it makes explicit the relationship between PA investments, the forms of value they generate, and who or what captures this value (see Figure 1.2). Put another way it enables a process of asset management based on four interlinked questions: 1) what forms of value does a PA (or PA system) generate and for whom? 2) what forms of value are wanted and who decides? 3) what forms of investment are needed in order that beneficiaries can capture this value? and 4) what forms of value could potentially be generated and for whom? Such questions will help strengthen the democratic accountability of PA policy, create links between other areas of policy (e.g. health, recreation, green development) and help ensure that organisations advocating and managing PAs are working for the wider public good.

***Figure 1.2:** Value creation*

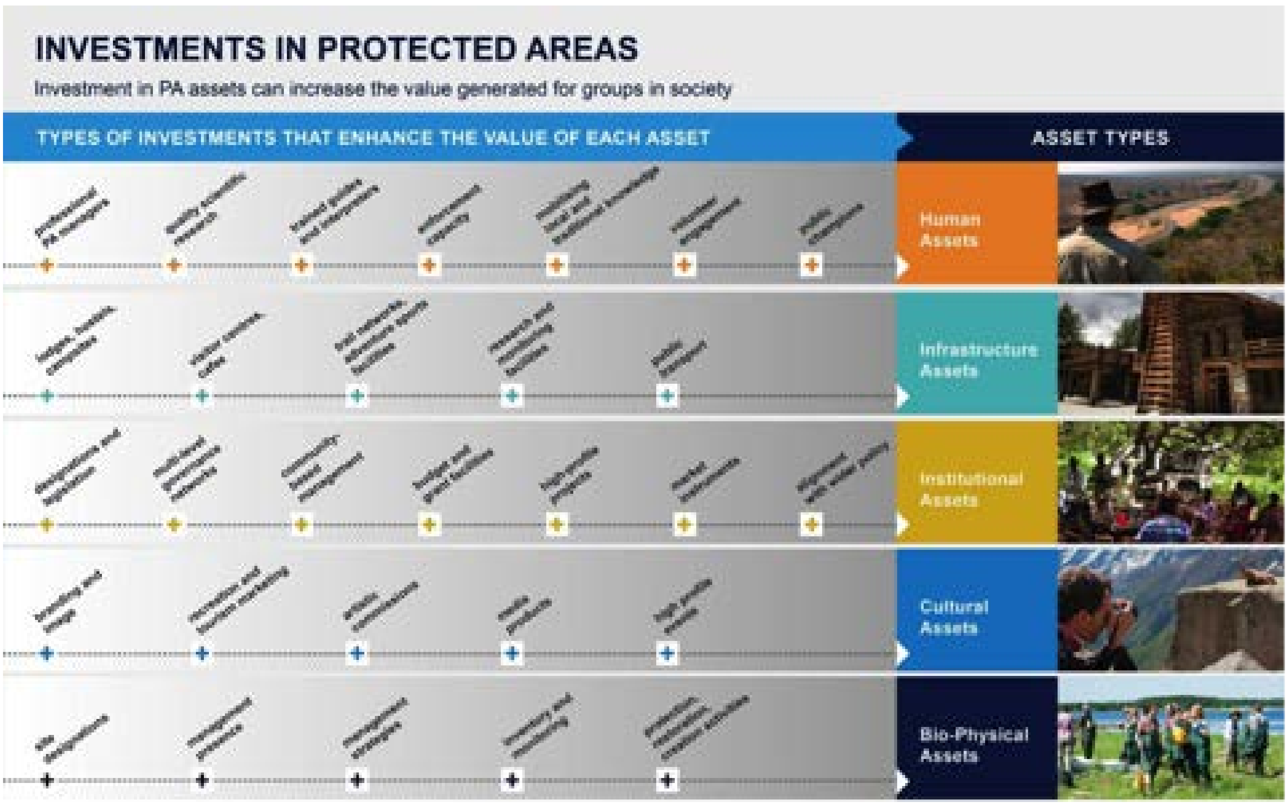
Different PAs and the 'package' of assets they embody will generate different sets of value that accrue to different domains in society. Once formalised our framework will allow politicians and publics to ask two key questions: First, are PA assets/investments (as a single site, portfolio or a network) optimal in terms of their spatial location, investment profile, and the forms of value they generate over time? Second, what forms of value generated by PAs are at risk and from which threats, and are they performing as well as they should in terms of value generation?

1.2 Attracting and assuring investment in protected areas

By revealing the range of value generated by PAs and the publics who capture, and could capture, this value our framework has the potential to attract a wider array of investments, from old, current, and new sources. Traditional sources of operational and capital funding, particularly from governments and conservation organisations, have been insufficient to assure PA assets and are unlikely to increase at the scale or pace required. Revealing the forms of value that accrue to different domains of society reinvigorates the case for investment to a broad range of actors. For instance, investment in outdoor recreation and natural beautification is sound and cost-effective for municipalities concerned with improving public health and wellbeing and attracting knowledge–based industries. The values of artistic and intellectual expression that accrue from PAs offer long-term returns for philanthropic funders interested in the arts and science. The values PAs generate in terms of identity and rural economic flows accrue as enhanced societal unity, tolerance, vision, and flexibility – i.e. societal resilience – that is of interest to governments and universal asset owners (for example, large pension funds).

Through the development of an asset framework we identify and specify types of investment that can develop and assure those types of PA asset which contribute to dynamic value creation in the long term (see Figure 1.3 below). Articulating a case for broader investment in PAs is one side of the coin. The other is assuring and reporting on the performance of such investment. By creating a model that formalises the asset-value-risk-investment relationship our asset framework supports and interacts with new work on PA verification standards and markets, specifically the VCA registry (VCA, 2015) and the IUCN Green list of Protected Areas.

Figure 1.3: *Investments in protected areas*



1.3 Value-at-Risk

Protected areas are coming under increasing pressure due to a number of threats, which are impairing their ability to create value (see Figure 1.4 below). Contemporary policy discourse focuses on threats to PAs arising from climate change, poaching, unsustainable exploitation, encroachment, fragmentation, breached ecological thresholds and so forth. Our framework extends this focus by revealing the importance of institutional, infrastructural, cultural, and human assets in protected area value generation. Risks to these include changes in land designation, regulatory change, jurisdictional tensions between ministries (e.g. mining and forestry), conflict between communities and authorities, shifts in visitor preferences, changing brand value, declining budgets and morale, corruption and excessive rent-seeking.

Further our framework posits that forms of engagement with PAs are integral to value creation and that these can also be at risk. Examples are commercial and policy barriers constraining recreational access, bureaucratic and funding barriers to scientific research and, to cite a very current example in Kenya, perceptions of health or other risks that undermine tourism. This broader and more holistic conception of value supports the development of a new generation of tools and procedures to identify, forecast, and manage risks to PA assets and investments.

1.4 What next?

Our typology of assets, values, beneficiaries, and risks (each with two subcategory levels) is intended as a framework that can be developed and applied for various purposes and in various contexts. We have tested its validity and utility by i) presenting the framework for review, critique, and adoption by communities engaged in protected area policy and finance; ii) applying a case study to assess the value generated and transferred during the lifespan (1974-2013) of a warden service in Greater Manchester, UK; and iii) developing a collective vision and strategy for a group of five reserves, each owned and managed by different conservation agencies on the Sefton Coast, UK.

Figure 1.4: *Value-at-Risk and risk management options*

Significantly, a full-scale developmental test of the framework is already underway in Brazil under a Brazilian National Council for Scientific and Technological Development CNPq-PVE grant (No: 400325/201) involving a partnership between the University of Oxford and the Federal University of Alagoas, Maceio. Brazil is an ideal country in which to test and develop our framework as a decision support tool. This is because Brazil has rich and varied biophysical assets, an extensive protected area system, but a growing population and economic demands are leading to calls for PADDD, particularly at lower administrative levels. However, Brazil is also investing strategically in its science capacity and has the most complete and accessible social, economic, and ecological data sources of any large and diverse country. This research includes an analysis of 200-plus Brazilian PA management plans (see below) and the development of next generation PA metrics using 'big data' and 'culturomics approaches'.

We are encouraged by the positive reception to the framework from professionals engaged with PA policy at various levels. A common reaction is that it chimes with new directions in organisational thinking that have yet to be structured in a framework. We are also encouraged by the quantity and quality of insights this can generate. For example, its application in Brazil has demonstrated the value of PA management plans as an institutional asset by revealing the somewhat narrow and formulaic process of their production and the potential to extend their scope beyond nature management to include value generation and investment planning. The application of our framework to the case of a recently disbanded warden service in Manchester showed that in the early decades of an urban conservation and restoration project a warden service generated multiple forms of value capture by multiple constituencies, but as these were 'normalised' in local society the value was no longer needed (e.g. presence related to safety) or were taken up by other actors (e.g. NGOs generating educational value). Lastly, the interaction between our framework and culturomics approaches has inspired the creation of new indices of PA saliency that act as a proxy for investor risk and/or rallying power against threats.

Earlier in this research we explored the merits of systematising the framework design using multiple categories. Based on our test applications we have backtracked from this approach realising that the basic framework design supports a situated, case-by-case approach to PA management. Because much of the value generated by PAs is relational in nature – it is contingent on sociopolitical context and era – it works best when the basic framework architecture is extended and applied for particular purposes. Our experience so far is that this is not particularly difficult to do and we believe that its application over time will build up a body of applied expertise and pratices suited to different situations and purposes.

Section 2a: Brazil

Authors: Paul Jepson, Chiara Bragagnolo, Ricardo Coreia, and Richard J.Ladle

2.1 Introduction

Brazil has a complex, volatile and unique national identity. Brazilian society and culture has emerged from the melding of Iberian colonists with native peoples and African slaves, its landscapes and environment, the aims and causes that have brought them together and their place in the world market. Brazil has a national ethnicity with its own unique characteristics, where people are unified by a common language and traditions, but distinguished by regional variations in environment, economy and immigration. This allows them to be variously known as quanchos (people of southern plains), cabaclos (of the Amazon), Nippo-Brazilians and so forth. Brazil is also characterised by marked social stratification that creates profound distance between privileged and governing elites and the mass of the population, such that social differences may frequently be even more difficult to bridge than racial differences (Ribeiro, 2000).

This report seeks to frame Brazilian protected areas as a spatial asset class – areas that have been designated to protect and invest in natural assets in order to generate forms of value (benefits or returns) for society and/or groups therein. In the context of protected areas, the concept of asset does not immediately align with Brazilian policy frames: Brazilian protected area officials typically equate the term with the concept of national patrimony – something that is valued and passed down from previous generations. Connecting the concepts of national patrimony with PA assets is crucial to this report and requires answers to several key questions, such as: who decides what is natural patrimony or a natural asset? what is to be valued, protected and passed down? what groups within society benefit from decisions on such questions and how do they affect investment? how do all these arise and how do they affect PA resilience?

2.2 Brazilian Protected Areas

Brazil has well-developed protected area institutions and a reserve system that comprises some 1,940 reserves (strictly protected and sustainable use) totalling approximately 1,513,828km (MMA, 2015). Brazil currently has about 17% of its territory conserved by PAs, but this proportion is not evenly distributed across biomes. For example, the Caatinga (semi-arid) and Pantanal (wetland) biome have less than 8% and 5% respectively of the land areas conserved (Ribeiro, 2000).

While the first Brazilian National Park (Itatiaia) was designated in 1937, investments in institutions and expanding the PA estate are relatively recent: starting in the late 1980s, gaining momentum after the 1990 Rio Earth Summit, and consolidated in the 2000 Forest Act (Law 9.985, July 19, 2000). This law was aligned with the government's restructuring of its environmental institutions. In 1989 the Brazilian Institute for Environment and Natural Resources (IBAMA) united two separate agencies with PA competencies. It is located within the Ministry of Environment and overseen by a deliberative body with civil society representation called the National Council for the Environment. In 2007 the Chico Mendes Institute for Conservation (ICMBio) was created, representing the executive board of the Ministry of the Environment, working exclusively to govern and manage PAs.

2.3 Landmarks in the creation of Brazilian protected area assets

In keeping with the idea of waves of protected area policy (see Section 1.1), differing protected area agendas have attained high-level policy attention in Brazil on at least six occasions.

1935-1940: The creation of foundational national parks, inspired by the Yellowstone model and covering extraordinary landscapes, namely Itatiaia (1937), Iguaçu (1939) and Serra dos Orgãos (1939).

1960s: First Forest Code and creation of the Brazilian Forest Development Institute (IBDF), a section of the Ministry of Agriculture with forest and PA competencies.

1970s: Creation of ecological stations to generate scientific knowledge for environmental management.

1970s: Creation of International Reserves following global initiatives (Biosphere Reserves, Ramsar Sites and Natural World Heritage Sites).

1980s: The Brazilian constitution recognised the establishment of protected areas as a means to preserve Brazilian diversity (cultural and biological).

1985: Chico Mendes rubber tapper's movement and creation of extractive reserves pioneered in Acre State.

1989: Creation of the Brazilian Institute for Environment and Natural Resources (IBAMA).

1992: Rio Earth Summit and signing of the Convention on Biological Diversity.

1988-2000: Development of the National Protected Areas System (SNUC) led by FunNature. This recognised a category of private protected areas known as Private Patrimony Reserves (RPPNs).

1992: Establishment of the Brazilian Biodiversity Fund (Funbio).

1990s: Development of sustainable use reserve concept, pioneered by Jose Marcio Ayres in Mamiraua.

1990s: Demarcation of indigenous lands under leadership of FUNAI (National Indian Foundation).

2002: Passing of the Forest Act which consolidated these initiatives.

2002-present: Amazon Protected Areas programme (ARPA) launched at the 2002 Johannesburg World Summit on Sustainable Development.

The Forest Act specifies the objectives of 12 protected areas grouped into Integral Protection Units (IPUs) and Sustainable Use Units (SUUs). This reflects two dominant policy worldviews within Brazil on the purpose of protected areas: the first emphasises intrinsic values and the preservation of biota without human intervention.

It is biocentric in orientation and associated with the traditional conservation biology perspective on conservation. The second worldview emphasises instrumental values and the sustainable use and management of natural landscapes. It is more anthropocentric in character and associated with socio-environmental perspectives on conservation (IS, 2015). The development of SUUs encouraged the designation of protected areas by state-level administrations.

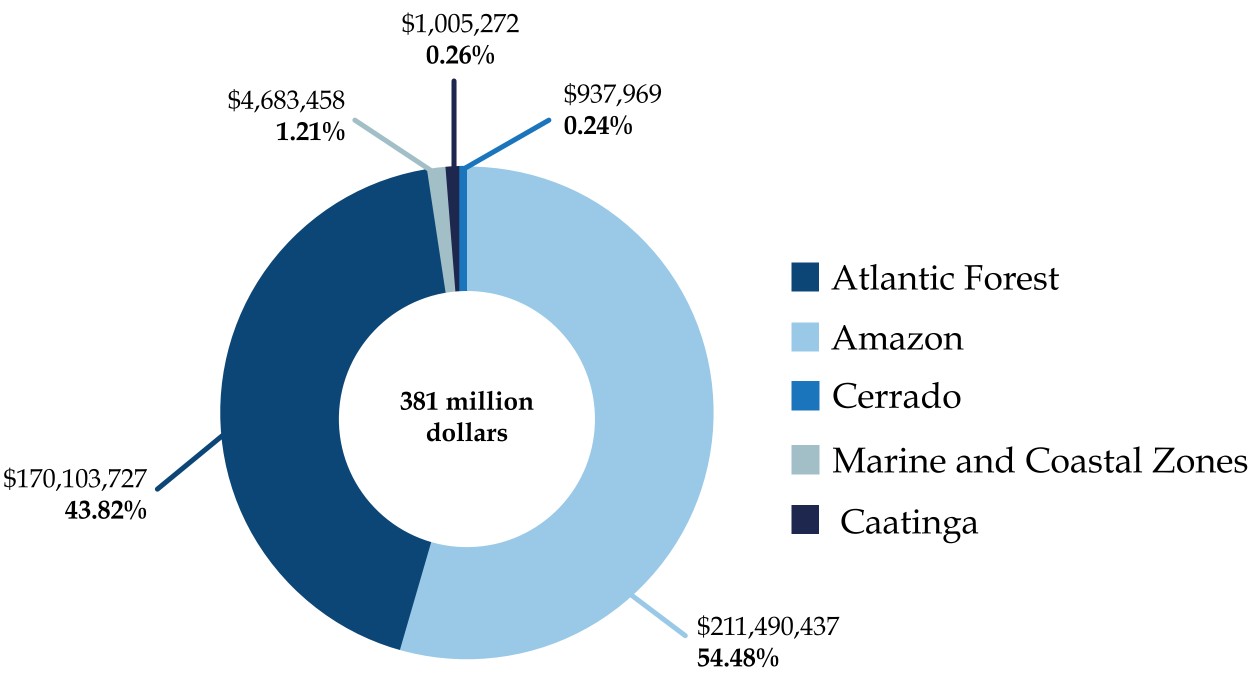
2.4 Investments in PA assets

In 1996 the Brazilian Ministry of Environment, with financing from the Global Environmental Facility (GEF), established the Brazilian Biodiversity Fund (Funbio) as an institutional asset tasked with developing a financial mechanism to invest in the expansion, consolidation and management of the PA network. Funbio has developed both the capacity and financial control systems (e.g. Cerebro) to manage large-scale donor projects (it became the 15th accredited GEF project agency in 2014) and created new financial instruments such as the Atlantic Forest Conservation Fund (AFCoF) and the Protected Area Bank. It provides services to funders and generates sector knowledge and analysis (Monterio, 2014).

Direct government investment in protected areas is relatively limited: in 2014 just 0.33% of the federal budget was allocated to the Ministry of Environment and only 18.5% of this went to ICMBio. The budget of ICMBio has fallen in real terms since 2010 and stagnated at critical levels. Moreover, the ongoing Lavo Jato (carwash) corruption scandal is expected to reduce the overall federal budget in the coming years, with sectors considered non-essential, such as the environment, likely to face further cuts.

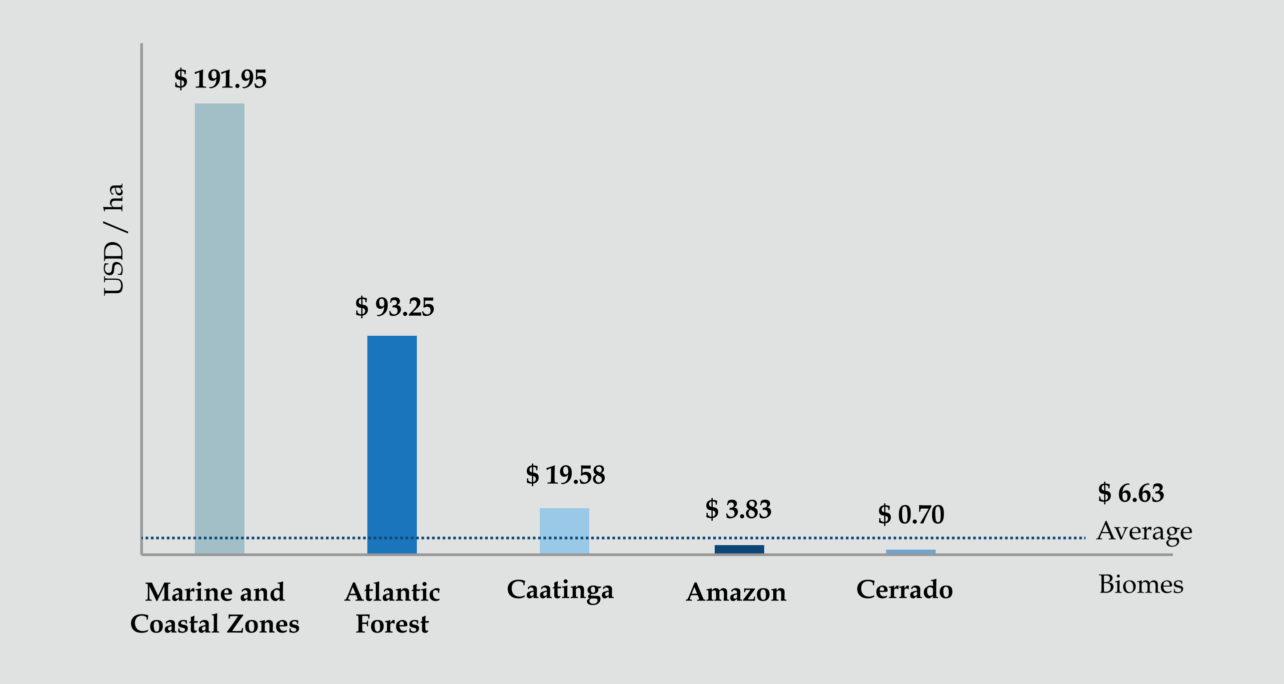
Since its creation Funbio has mobilised US$477.6 million of which US$388 million (81%) has been invested in protected areas. Most of this amount has been invested in the Amazon and Atlantic Forest Biomes (Figure 2.1). The Atlantic Forest has received significantly higher investment per hectare than other terrestrial biomes.

Figure 2.1: *Amounts contracted to fund protected areas, per biome (in US dollars)*



Funbio has managed and mobilised investments from five principle sources (Figure 2.2). Funds from bilateral and multilateral donors account for 45% of total investment and are almost exclusively targeted to the Amazon biome. There have been two major multilateral investments in Brazilian PA assets: a US$428 million investment under the Pilot Programme to Conserve the Brazilian Rain Forest (PPG7) operational 1992-2005, and the ongoing US$220 million investment in the Amazon Region Protected Areas Programme (ARPA) launched in 2002.

Figure 2.2: *Average investment in each biome, per hectare (in US dollars)*

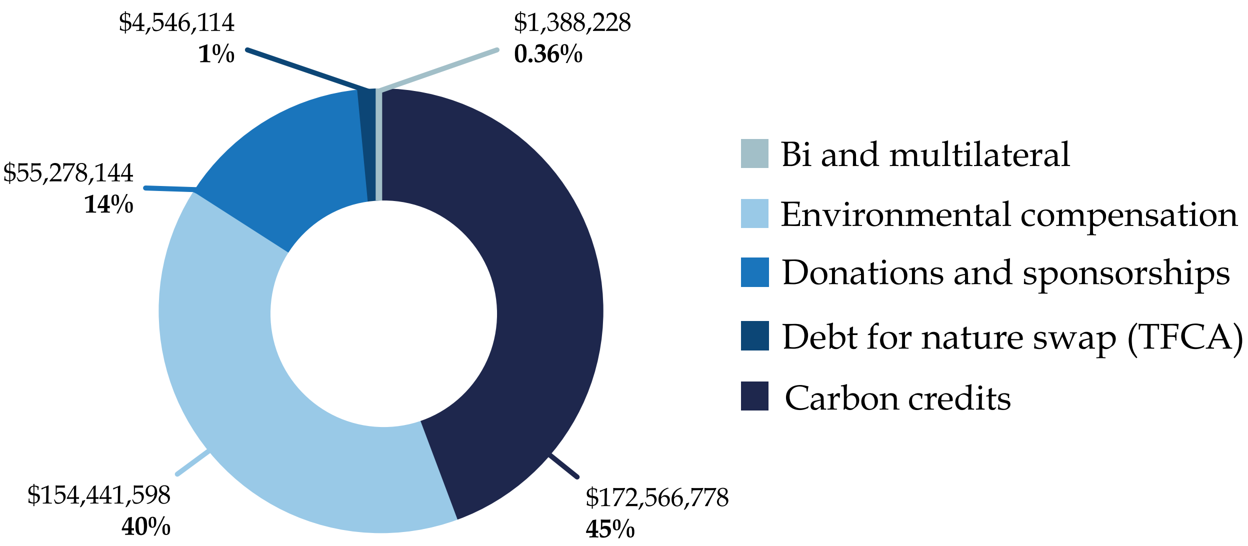


PPG7 was financed by the governments of Germany, the Netherlands, Italy, France, Japan, Canada, UK and US, the European Commission and the Brazilian government. They contributed to a Rain Forest Trust Fund (RFT) managed and executed by the World Bank. It represented an investment in: i) the demarcation of 45.4 million hectares of indigenous lands and 2.1 million hectares of extractive reserves; ii) the adoption of forest certification in the Brazilian Amazon; and iii) the development of associated institutional assets relating to participatory/community reserve management and civil society networks (Funbio, 2015).

ARPA is the largest current investment in tropical forest protected areas globally, and like PPG7 is intended to expand and strengthen the National System of Protected Areas (SNUC) in the Brazilian Amazon. This investment aims to: i) create 283,000km2 of new protected areas for strict conservation use and 89,000km2 of new sustainable use reserves; ii) improve the management of 125,000km2 of existing parks; iii) create a draw-down endowment fund capitalised at US$215 million (in 2014) to provide long-term financing of these reserves, and; iv) develop decision-support PA planning studies and tools that enable assessments of management effectiveness, progress and investment, (ARPA, 2015).

Brazil has pioneered environmental fiscal transfer schemes (EFT) to reward local government efforts toward sustainable development. **The Ecological Value Added Tax** (ICMS-Ecológico) was designed and introduced in 1991 to compensate municipalities for land use restrictions. It now accounts for 40% of total investments and has great untapped potential. The amounts included in Funbio's figures (Figure 2.3) are only from Rio de Janeiro state. ICMS-E has now been adopted by 18 states (OECO, 2015). In 2009, 11 states received US$142 million (R$446 million) via this instrument of which (US$129.3 million (R$406 million) was related to PA criteria. The instrument has made PAs part of the agenda of public administration, but its impact in terms of investing in reserve expansion and/or management is less clear: only four states have increased the size of their PA estates, however this may be because there are limited opportunities for other states. Furthermore, the instrument has two shortcomings: 1) the National Taxation Code means that taxes are not tied to specific expenditures. Thus, ICMS-E revenues are only earmarked for PAs if complementary legislation is passed; 2) the instrument is 'rival', meaning that revenues are diluted as more municipalities in a state apply and a larger area is protected (OCEO, 2015).

Figure 2.3: *Funds raised for protected areas (in US dollars per mechanism)*



Three additional financial instruments should be mentioned to complete this overview of the Brazilian PA asset investment landscape. First, the 2000 Forest Code exempts owners of RPPNs (Private Patrimony Reserves) from Rural Territory Tax on the area of land designated. Once designated, RPPNs must be protected in perpetuity (Ladle et al, 2014). This tax exemption therefore represents a government investment in private protected areas. Second, Funbio has created a range of PA investment funds that receive contributions from conduct adjustment agreements (with the perpetrators of environmental damage, environmental administrative fines and philanthropic, corporate and other sponsorship). Examples include Fauna Brazil Portfolio and Adopt a Park (Funbio, 2015). Finally, Funbio has created a mechanism that provides PAs with a bank accounts and credit cards to enable park managers to purchase the goods and services needed for everyday operations. This is currently available for PAs in the ARPA programme and the funds available are based on expenses authorised in the plan of operation and by donors.

The diverse mechanisms outlined above provide direct support to create and consolidate protected areas and indirect support to help PAs deliver on their purpose. Direct support finances activities include planning studies, management plans, establishment of management councils, infrastructure, boundary demarcation and resolution of land tenure issues. Indirect support finances research, species monitoring and management, capacity building of reserve managers, environmental education and community projects.

2.5 Forms of value generated and beneficiaries

Brazil's investment in PA assets is relatively recent: significant and systematic expansion of federal and state reserves commenced only 20 years ago. As such, most investment has focused on the planning and designation of PAs and establishing the institutional infrastructure to support the administration and financing of the largest national reserve system in the world, representing 12.4% of the area of protected areas globally (Bernard et al, 2014).

This has generated forms of political, technocratic and organisational value. By encoding principles of the international biodiversity regime in its Forest Code (e.g. the representation principle) and actively expanding its PA estate, Brazil assured its environmental leadership status in this regime and the diplomatic value accrued from hosting the 1992 Rio Earth Summit and Rio+20 summits. At the same time, Brazil's active participation legitimates the international biodiversity regime and contributes to data completeness and the meeting of international biodiversity targets. This contributed value for IGOs such as the IUCN and WCMC and international NGOs such as WWF, CI, and TNC whose business models are founded on contracts and project overheads associated with technical data management and PA planning funding flows. The bilateral and multilateral aid flows have supported the expansion of Brazilian institutes and NGOs, such as IMAZON, FunBio, SOS Mata Atlantica, and in so doing it has created professional jobs and careers for educated Brazilians. Furthermore, the emphasis on protected biodiversity assets has generated international status and prestige for Brazilian biodiversity scientists and foreign scientists working in Brazil. As previously indicated, registered municipalities also gain some value from ICMS-Ecológico.

In addition to the forms of value mentioned above, Sustainable Use PAs generate value for local and indigenous people living within and/or dependent on them for their livelihoods. The most significant value generated is security of traditional livelihoods and ways of life by legally excluding commercial agriculture and extraction forces from their lands. Furthermore, many SSUs (and in particular Environmental Protection Units) have advisory councils. Such institutional assets promote better governance in the area through bringing together local and external expertise and generating wider policy attention for the needs and perspectives of traditional peoples.

The Forest Act specifies the forms of value that each protected area should or can generate (Table 2.1). These are narrowly specified in legislation and the PA management plan is specified as the instrument to identify and cost investments that will generate broader forms of value such as outdoor recreation. As of July 2015, management plans have been prepared for 204 PAs (Icmbio, 2015) and a study of the investment strategies these represent is underway. Preliminary results suggest strong adherence to the limited human impact ethos of the legislation and minimal investment in assets that would generate value for public visitation, outdoor recreation, cultural events or regional identity-building. This may in part be due to severe constraints on government PA budgets.

The monetary value generated from Brazilian PAs mainly stems from forest products, public use (tourism and visitation), carbon sequestration, water provision and distribution of tax revenues.
Brazilian PAs also generate tourism value although data on this aspect is limited. Between 2007 and 2011, 20% of international tourists visiting Brazil said that their reason for visiting was nature, ecotourism or adventure. However, it is expected that relatively few iconic reserves generate international tourism value. For example, in 2012 Tijuca in the middle of Rio de Janeiro attracted 2.5 million visitors, Foz de Iguaçu (Iguassu Falls) 1.5 million but the third ranking national park, Brasilia attracted just 318,000 (Ecotourism, 2015).

Table 2.1: *Forms of value that each protected area should or can generate, as specified in the Forest Act*

PA	Category Name	Objective	Value beneficiaries	
	Integral Protection Units (IPU)	Preserve nature. Only indirect use of its natural resources, except as specified in management plan.		Public Domain
I	Ecological Station	Preserve nature & conduct low impact scientific research.	State: knowledge to support decision making. Scientists: research projects Higher Education: educational visits.	Public Domain
II	Biological Reserve	Preserve nature without direct human interference or environmental changes, except for recovery of altered ecosystems.	Scientists: research projects	Public Domain
III	National Park/State Park/ Parque Municipal Natural	Preservation of nature & scenic beauty, enabling scientific research, educational activities, environmental interpretation, and recreation.	Scientists: research projects Higher Education: educational visits General public: out-door recreation Tourism sector	Public Domain
IV	Natural Monument	Preserve rare natural sites	(As above)	Private or Public Domain
V	Wildlife Refuge	Protect natural conditions for maintenance of populations or resident and migratory fauna and flora.	(As above)	Private or Public Domain
	Sustainable Use Units (SUU)	Nature conservation with compatible sustainable use of a portion of its natural resources.		

(Continued on next page)

Table 2.1: Continued from previous page

PA	Category Name	Objective	Value beneficiaries	
I	Environmental Protection Area	Extensive natural areas with a degree of human occupation protected to conserve natural and cultural attributes, lifestyles and ensure sustainable use of natural resources.	Governing body sets use limits and conditions for scientific and research public viewing	Private or Public Domain
II	Area of Relevant Ecological Interest	Small uninhabited area of extraordinary value where regulation of human use is necessary for conservation objectives.	As for parks and monuments	Private or Public Domain
III	National Forest	Sustainable forest management and research.	State: sustainable forest policy & science Forestry sector : commercial operations Forest scientists: research projects Publics: educational & recreational visits.	Public Domain
IV	Extractive Reserve	Protect livelihoods and the culture of traditional populations based on sustainable use of natural resources in the unit.	Traditional populations: security from other land/resource interests, governance institutions. Scientists: researchers Public: visitation	Public Domain
V	Wildlife (Fauna) Reserve	A natural area suitable for technical and scientific studies on the sustainable economic management of wildlife resources.	Scientists: research projects Public visitation	Tenure & Public Domain
VI	Sustainable Development Reserve	Preserve nature and improve life quality of natural resource-based traditional communities along with their r knowledge and techniques of environmental management.	Traditional peoples: Security of access to resources, development assistance. Scientists: research projects Public visitation	Public domain
VII	Private Natural Heritage Reserve	To conserve biological diversity	State: execution of policies Land owners: reduced land tax access to grants.	

PA creation and maintenance sequesters at least 2.8 billion tons of carbon, conservatively estimated to have a value of US$48.3 billion. In addition, the estimated value of the carbon stock held by these areas is from US$1.46 billion to US$2.92 billion/year.

Estimated timber extractable from national and state forests of the Amazon has the potential of generating annually between R$1.2 to 2.2 billion, more than all native timber currently extracted in the country.

Moreover, according to the estimates of future tourist flow (about 13.7 million people by 2016), between R$ 1.6 and 1.8 billion per year could be generated through visits to the 67 existing Brazilian national parks. These financial resources could help to both maintain the parks' infrastructure and support regional development (MMA, 2015).

Scientific research is a prime objective of several PA categories, especially ecological stations that appeared in the late 1970s. These were specifically created to generate data for use in national policies and technologies to control and manage the environment (Nogueira-Neto & Carvalho, 1979). Brazil has strategically invested in developing its science capacity, but PA assets have so far contributed relatively little to this.

2.6 Risks to Brazilian PA assets

Brazilian PA assets are at risk from a number of social and economic pressures, especially in the agribusiness and energy sectors. The former, primarily represented by cattle rearing, soy and sugar cane production, often directly competes for land with PAs. Brazil's energy demand is predicted to grow by 72% by 2035 (BP, 2015). Brazil is very dependent on hydroelectricity and the plans for hydroelectric plants on all large rivers in Amazonia places PA assets at risk.

Limitations in PA management capacity mean that biophysical assets within PAs are at risk from informal and illegal extraction such as logging, hunting and small-scale agriculture. A 2007 study of the quality of management in federal Conservation Units (CUs) found that management was highly effective in 13% of CUs, moderately effective in 36% and mostly ineffective in the remaining 41% (MMA, 2015).

PA assets are also at risk from conflicts with economic development interests. A 2014 study identified 93 PADDD events (Protected Area Downsizing, Degazettment and Downgrading) associated with federal and state reserves in the period 1981-2012 (Bernard et al, 2014). Of these 93 events, 69 occurred in IPUs and 24 in SUUs and the study found a marked increase since 2001 – before this there had only been six. PADDD in Brazil is occurring in an ad hoc manner based on individual presidential (for federal CUs) or state governor decrees. There is undoubtedly a need for some adjustments to the SPCU systems but these are being made without strategic assessment.

One of the greatest risks to PA assets is diminishing support for PAs in the Brazilian Congress: the political forces needed to assure PA assets are lacking, and the PA ideal generally lacks wider social support. We believe this is in part due to the dominant environmentalist/preservationist ethos of the legislation, which generates value for technocrats and traditional peoples but not the wider Brazilian public living in cities or towns. Although Brazilians view 'big scenery' as being typical of their country, the legislation does not draw attention to this view.

One estimate suggests that assuring Brazilian PA assets will require direct support of US$900 million a year with an additional US$450 million a year investment in PAs if they are to be effective. This represents a six to eightfold increase on the present situation. To attract this level of investment we suggest that Brazilian PA policy needs to move into a new, post-establishment phase. One that strategically assesses the potential of PA assets to generate forms of value that are meaningful to, and can be enjoyed by, the wider Brazilian public. They should interact with cultural and recreational trends in society to enhance quality of life, and provide new opportunities for enterprise and the Brazilian economy. The asset framework outlined in this document offers one approach towards the development of Brazilian PA policy.

Table 2.2: *Information is provided on a selection of current and future threats facing the PA network of Brazil with specific examples given where possible. A status is given to the threat depending on seriousness, i.e. green is minor, orange moderate and red severe. The timescale of the threat is also indicated*

Threat	Overview / examples	Status / timescale
1. Physical climate change	Climate change, including increased temperatures in Brazil, is leading to range shifts. Consequentially, some protected areas will gain new complement of species while others will gradually lose species (Hole et al. 2009). Henece, there is a need to ensure PA network as a whole is resilient.	Orange Long- & short-term
2. Illegal wildlife trade and poaching	Brazil has a very strict law punishing wildlife crimes. However, keeping animals in captivity and hunting are still deeply rooted in local culture (Alves et al, 2013). The commercialisation and use of wildlife has huge implications for conservation and management of PAs, especially in north and northeastern Brazil. A recent study showed that only 1% of the total fines issued for crimes against wildlife in Pernambuco northeastern state was paid. Such inefficiency in reinvesting an important source of funding for conservation poses a threat to the value and persistence of Brazil's protected areas (Bernard et al, 2014).	Red Long-term and increasing in frequency and intensity
3. Land encroachment: Deforestation	Brazil is a country where demand for food and fuel leads to massive forest degradation and deforestation (Fisher et al 2011). This phenomenon is particularly acute in the so called 'arc-of-deforestation' surrounding the Amazon region. The main causes are timber extraction, new roads, intensive agriculture (i.e. soybean cultivation), predatory logging, burning, and expansion of cattle ranching. According to Global Forest Watch, Brazil was the country with the second greatest tree cover loss between 2001 and 2013, with a net tree cover gain of 7,586,752 against a net loss of 35,757,707ha. Deforestation rates in Brazilian PAs are much lower and are monitored under the PRODES programme that publishes up-to-date data on changes in the rate of deforestation. Between August 2009 and January 2011, deforestation in Protected Areas (including indigenous lands) totalled 382 km2, the equivalent to 16.3% of the total deforestation that occurred in the Brazilian Amazon. In 2011, the forestry sector contributed US$22.5 billion to the Brazilian economy, which is approximately 1.1% of the GDP; 772,000 people are directly employed by the forestry sector (FAO, 2014).	Orange Continuous and Long-term impact

Table 2.2: Continued from previous page

Threat	Overview / examples	Status / timescale
4. Extractive industries	Mining has grown from 1.6% of GDP in 2000 to 4.1% in 2011. And production is expected to triple by 2030 (Ferreira et al, 2014). There is great concern about the proposed new Mining Code, calling for 10% of PAs (including Integral Protection Units) to open for mining concessions, and discouraging the creation of new PAs in areas of high mineral or hydropower potential. Some 34,117 km2 of strictly protected areas in the Amazon (8.3% of total area) lie in areas of registered interest for mining (Ferreira et al, 2014).	Red Short term impacts from land-clearing during construction. Longer term impacts from pollution, infrastructure, lost tourism
5. Unsuitable infrastructure development	Rising demands for hydropower are putting great pressures on Brazilian PAs. Hydropower accounts for 77% of Brazil's energy supply, while 70% of national potential remains untapped (Ferreira et al, 2014). Of Brazil's total untapped hydropower potential of around 180,000MW, about 80,000MW lies in protected regions, mostly indigenous territories, for which there are no development plans. Perhaps the most controversial hydroelectric project facing Brazil today is the Belo Monte Dam, the world's third-biggest hydroelectric project. It would divert the flow of the Xingu River, flooding an area of 516 km2 in the Amazon basin (most of which is rainforest), displacing over 20,000 people, and threatening the survival of indigenous tribes that depend on the river. In addition road infrastructure development is an increasing problem. For example, a road crossing the most visited Brazilian National Park (Iguaçu National Park) was closed in 2001 by an order of the Federal Court, as threatening the integrity of the park and national security. Its proposed reopening would set a dangerous precedent, weakening the National Protected Areas System and the protection of biodiversity and ecosystem services in the country. The Iguaçu National Park is one of the most important preserved remnants of the Atlantic Forest. The road would cut through a core area, further endangering the movement of important species such as the jaguar.	Orange Short-term negative impacts from destruction and degradation of habitat Long-term negative impacts from habitat degradation, pollution Long-term positive impacts could include better access to certain PAs
6. Regulatory change: Protected Area Downgrading, Downsizing, and Degazettement (PADDD)	The online resource that tracks PADDD around the world, PADDDtracker.org, identified 95 PADDD events across the Brazilian protected area network. PADDD phenomena in Brazil are also receiving the attention of scholars (Araújo & Barreto, 2010; Bernard et al, 2014; de Marques & Peres, 2014) who are concerned about the recent shifts of Brazilian conservation policy in favour of development projects (i.e. hydroelectric dams and roads), agricultural land conversion and local demands to relax restrictions on land and/or natural resource use, even when in direct conflict with established conservation units (de Marques & Peres, 2014).	Orange Long-term: This is a real and continued risk and undermines management efficiency

Table 2.2: *Continued from previous page*

Threat	Overview / examples	Status / timescale
6. Regulatory change: Lack of IUCN management criteria	Not relevant to Brazil, it has its own well developed system (see above).	Green Becomes more important when other threats are present as management criteria determine strength of resources for maintaining PA integrity.
7. Jurisdictional tensions between ministries (e.g. mining and forestry)	Recent legislative reviews	
8. Conflict between communities and authorities: poverty land tenure 8. Corruption	Despite its rapid economic growth, the Human Development Index (HDI) for 2013 still ranks Brazil in 79th position (Human Development Report, 2014). There is also a great variability within the country, since the northern and northeast regions are the least developed. Historically, an internal migration from rural areas of northeast regions to southern cities (i.e. São Paulo, Rio de Janeiro, etc.) was driven by socioeconomic and environmental factors (drought, rural market, etc.). Rural and traditional communities in particular are highly dependent on natural resources, and affected by any restrictions on access to these resources. Local communities are more included in protected area management than before but conflict over the establishment of reserves still arises and threatens the natural state of Brazilian protected areas. Land tenure in public PAs is another critical issue in Brazil as private owners having land inside a public PA need to be paid by the government (this process is called land regularisation). The process is frequently slow and extremely bureaucratic and obtaining data is not straightforward. In 2013, the chief of ICMBio declared that it would needed approximatively about R$ 12 billion (about US$3 billion) to buy the private land of the 312 federal PAs, the equivalent of building 12 Maracanã stadiums). Example: The Chapada Diamantina National Park was created in 1985. Today, more than 30% of the land is still privately owned and about 15% is legally contested.	Orange Long-term.

Table 2.2: Continued from previous page

Threat	Overview / examples	Status / timescale
9. Corruption	Due to the national politics and corruption scandal in 2014 (above) state expenditure has been severely cut and this impacts on PAs.	Green Intermittent threat, difficult to predict.
10. Local conflicts and geopolitics	Some states see PAs as the imposition of federal jurisdiction on their lands. This may lead to calls for PADDD. The so called 'ruralistas', a caucus of landowner and agribusiness interests in the Senate is generally antagonistic to protected area and indigenous rights policy.	Green Short-term, likely to increase
11. Breached ecological thresholds	-	-
12. Shifts in visitor preferences	-	Green Long-term, potentially increased threat in the future

Section 2b: Tanzania

Author: Harriet Milligan

2.7 Status and history

Tanzania has a land surface area of 947,253km2, with an estimated population of 49.25 million in 2013 (World Bank, 2015), and population growth estimated at 3%, currently higher than neighbouring Kenya and Mozambique. Tanzania has a predominantly rural population and natural systems are integral in indigenous ways of life; communities rely heavily on local resources.

Tanzania's protected area network consists of 626 areas making up 26% of its terrestrial and marine area (Protected Planet, 2015). Tanzania is currently exceeding the target of 17% set by the Convention on Biological Diversity (1992) for the percentage of land protected, but is falling short of the marine area target of 10% (Table 2.3). Tanzania's natural features are highly diverse and its PAs include seascapes, large savannahs, forests and mountain ecosystems. Many provide refuges for threatened and/or endemic species and levels of biodiversity are high. Selous was one of the first protected areas, designated as a hunting reserve in 1905. The first designated national park in Tanzania was the Serengeti in 1937 and there has been a steady increase in area protected since 1950s. Four PAs are UNESCO World Heritage Sites (Kilimanjaro National Park, Serengeti National Park, Ngorongoro Conservation Area, and Selous Game Reserve). After independence in 1961 President Julius Nyerere placed great emphasis on PAs as a means for economic development. Since then the PA network has increased in size by ~150% incorporating 140 new PAs (WDPA, 2015). Conservation in Tanzania is governed by the Wildlife Conservation Act of 1974.

***Table 2.3:** Summary information for Tanzania's PA network (terrestrial and marine area). Source: World Database of Protected Areas (protectedplanet.org)*

	Terrestrial	Marine
Area protected km2 (%)	303,316 (32%)	6,713 (2%)
Average area of PA km2	751.88	338.15
Number PAs	619	29
Number PAs with international designation	10	1
Number PAs with national designation + management categories	587	28
Number PAs without national management categories	481	23

2.8 Values

2.8.1 Biodiversity and conservation

The National Biodiversity Index (NBI) measures species richness by abundance for four terrestrial vertebrate classes and for vascular plants; vertebrates and plants are ranked equally. The index values range between 1.000 (maximum) and 0.000 (minimum). NBI adjusts for country size. Countries with a land area of less than 5,000 sq km are excluded.

Tanzania has high levels of native biodiversity, with a NBI of 0.657, and is home to a number of endemic species, as well as many of Africa's large mammal species. The PA network has inherent value by providing habitat for this rich biodiversity, and aesthetic and wellbeing value in the opportunity for visitors to experience natural landscapes and wildlife. The Eastern Arc Mountains in particular are a global biodiversity hotspot with high levels of irreplaceable and endemic species (Myers et al, 2000, Burgess et al, 2007). However this highly diverse area of Tanzania is a major gap in the global PA network (Rodrigues et al, 2004) and within Africa for many species of bird (Klerk et al, 2004), plant (Burgess et al, 2005) and mammal (Fjeldsa et al, 2004). In the last decade 27 vertebrate species new to science have been described in the Eastern Arc Mountains, leading to its proposed designation as a UNESCO World Heritage Site.

2.8.2 Living Planet Index (LPI) for species populations in Tanzania's protected areas

The LPI is a biodiversity indicator that measures trends in vertebrate species populations (Collen et al, 2009). The LPI for data available from Tanzania indicates there has been an average decline of 69% in vertebrate populations within protected areas (18 PAs) over the last 40 years (Figure 2.4). Abundance data for 156 populations of 45 species contribute to the index. Twenty species have no or positive change, 25 species have populations with negative change. When abundance change is averaged for the PAs for which we have data, the majority are performing poorly with respect to vertebrate populations (Figure 2.5) with most exhibiting average population declines.

Table 2.4: *Species with population declines (left) and increases (right) across PA network since 1973*

Populations declines across PA network since 1973	Population increases across PA network since 1973
Roan antelope (Hippotragus equinus) – 134% decline	Spotted hyena (Crocuta crocuta) – 193% increase
African elephant (Loxodonta africana) – 124% decline	Common eland (Taurotragus oryx) – 100% increase
Impala (Aepyceros melampus) – 95% decline	Side-striped jackal (Canis adustus) – 70% increase
Topi (Damaliscus korrigum) – 91% decline	Zebra (Equus burchellii) – 33% increase
Black-backed jackel (Canis mesomelas) 98% decline	

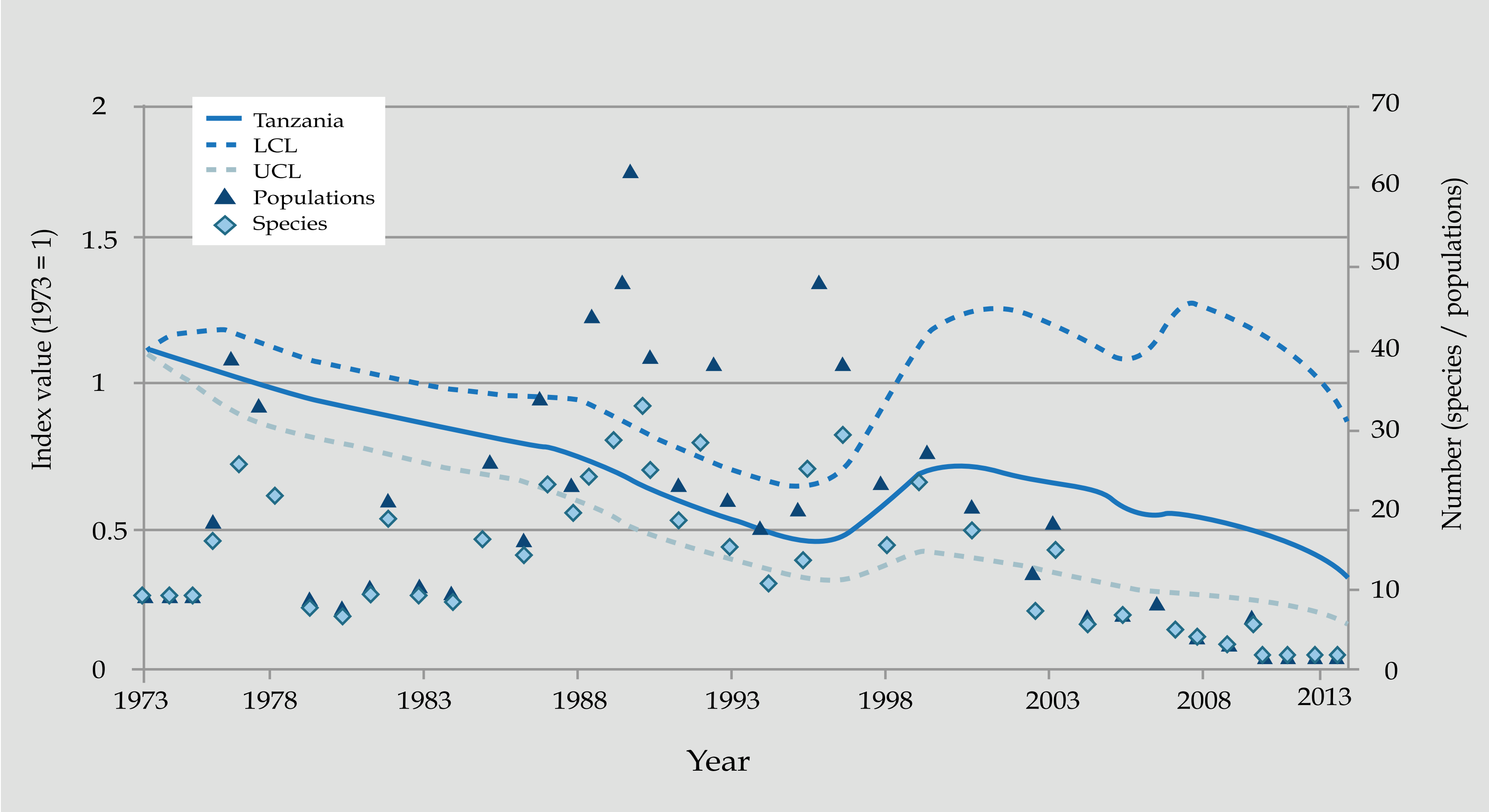


Figure 2.4: *The Living Planet Index (smoothed, plus upper and lower confidence intervals) for vertebrate populations located within PAs of Tanzania between 1973 and 2013. Number of species and populations that contribute to the index in a given year are indicated by pink diamonds and green triangles respectively. The LPI baseline is set at 1 in 1973. Trends below the line represent decreasing population abundance and trends above the line an increasing abundance*

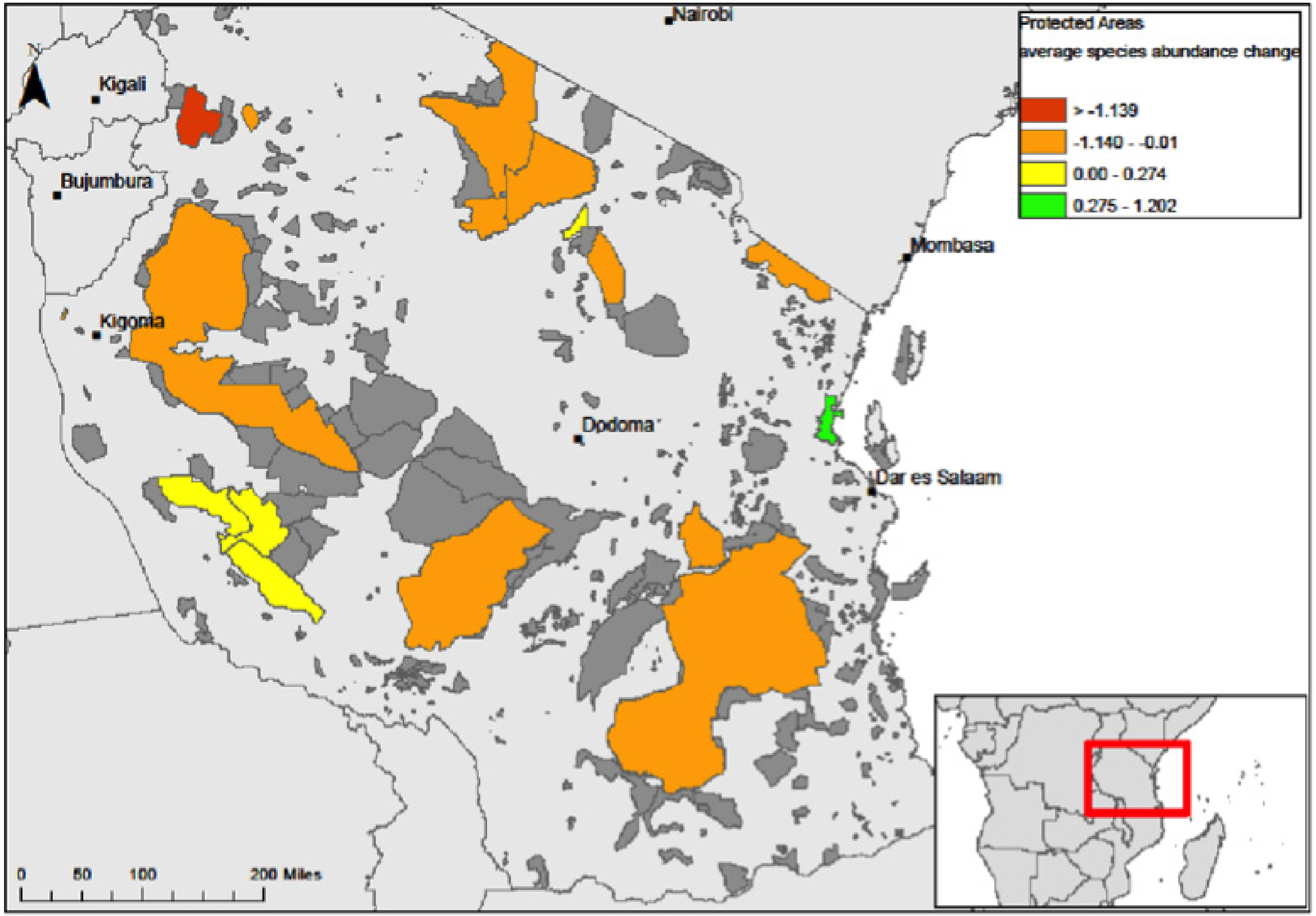
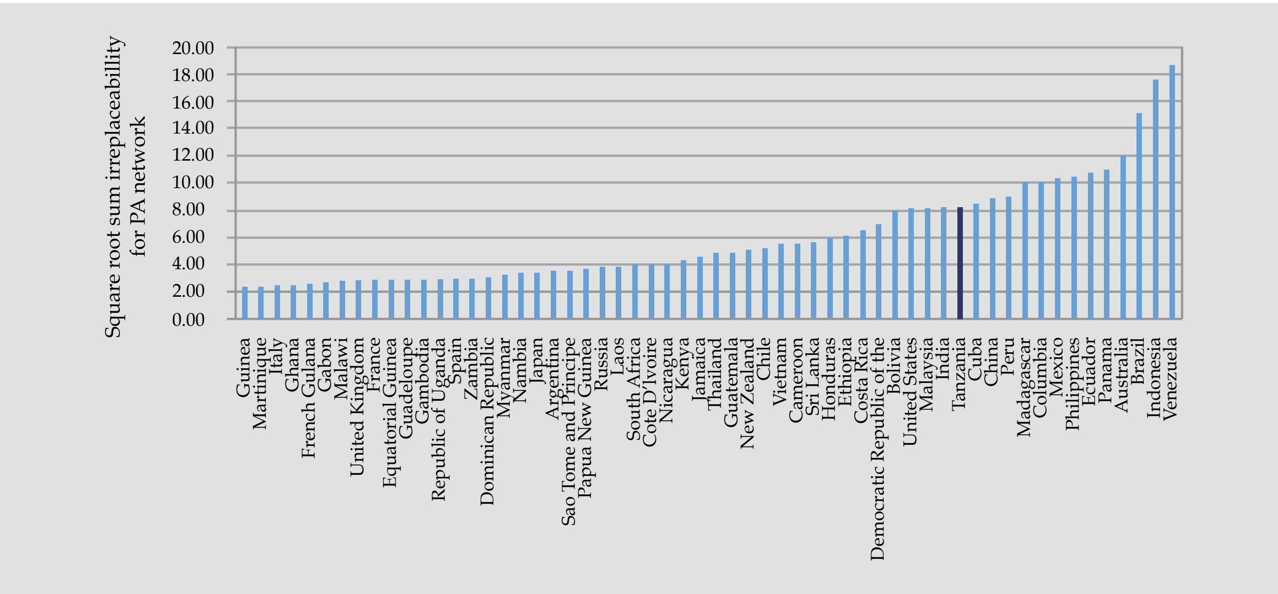


Figure 2.5: *PA network of Tanzania with PAs highlighted for which there is vertebrate population time series data (N = 18). Colours indicate the average population abundance change, with red and orange colours indicating declines and yellow and green increasing population numbers. Data is based on 156 populations, 45 species of birds, mammals and reptiles. Abundance change values indicate the percentage change from baseline of 1 set at 1970, e.g. -1.14 equals a 114% decline, whereas 0.274 equals a 27.4% increase*

2.8.3 Protected Area irreplaceability

Another indicator of the value of Tanzania's PA network is its irreplaceability for global biodiversity, in particular threatened vertebrates (Le Saout et al, 2013). The score considers the overall level of biodiversity a PA incorporates and the number of threatened and/or endemic species by overlap analysis of species distributions. The score is a general indicator of a PA's value relative to others in the network. In Tanzania, the irreplaceability of the PA network ranks 14th globally (Figure 2.6), and has steadily increased over time as more PAs are designated (Figure 2.7). Four of Tanzania's PAs are listed in the top 100 PAs for irreplaceability for threatened species (Table 2.5).

Figure 2.6: *Total irreplaceability score for the Tanzania PA network (minus 22 sites) in the context of the 58 top ranked countries (irreplaceability score > 5.0). Tanzania is highlighted in dark purple and placed 14th*



The species for which the top ten PAs of Tanzania are of irreplaceable global importance (i.e. distribution overlap with PA >5%) include 46 amphibians, 15 birds, and 18 mammals (Le Saout et al, 2013). This includes 46 threatened species and eight species found only within one PA (all amphibians). PAs that are attributed high irreplaceability due to overlap with many species of mammal, bird and amphibian (high species richness) do not have high overlap with high irreplaceability PAs due to overlap with threatened species distributions (high risk) (Figure 2.8 and Figure 2.9). This has implications for conservation management and in directing efforts to certain PAs based on conservation priorities.

Figure 2.7: *The cumulative terrestrial and marine area under protection (km2) is plotted on the primary y-axis (in red). This value is approximate and includes some overlap of sites. The cumulative irreplaceability scores of the PAs over time, as they are designated, is plotted on the secondary y-axis (in green). Some PAs lack designation date information (N = 258) and are included at the start (1905). Irreplaceability scores are missing for 22 PAs.*

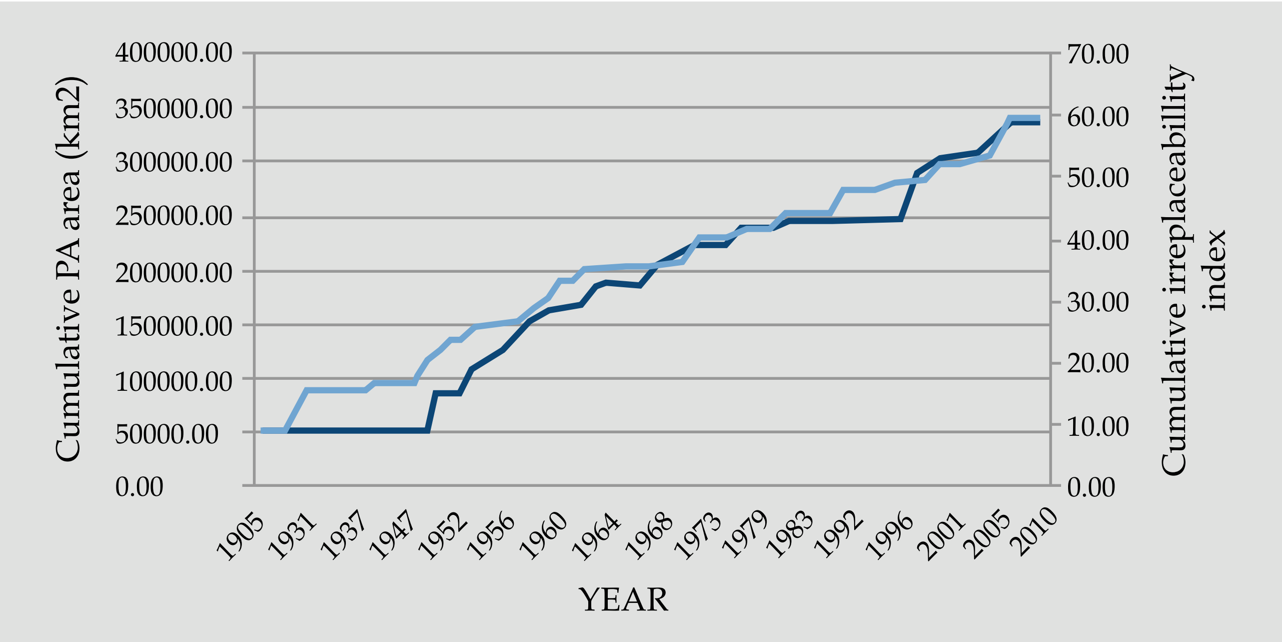


Table 2.5: *Summary of four of Tanzania's PAs that are ranked in the top 100 globally, based on their irreplaceability for threatened species of mammal, bird, reptile and amphibian*

Protected area (WDPAID)	Area (km2)	Irreplaceability score – All (rank)	Irreplaceability score (threatened)
Milindo Forest Reserve (301550)	86	3.04 (200)	3.02 (90)
Nguru South Forest Reserve (303492)	198	3.26 (185)	3.14 (85)
Udzungwa Mountains National Park (19297)	2095	3.89 (146)	3.52 (70)
West Kilombero Scarp Forest (301596)	1923	4.56 (106)	4.05 (51)

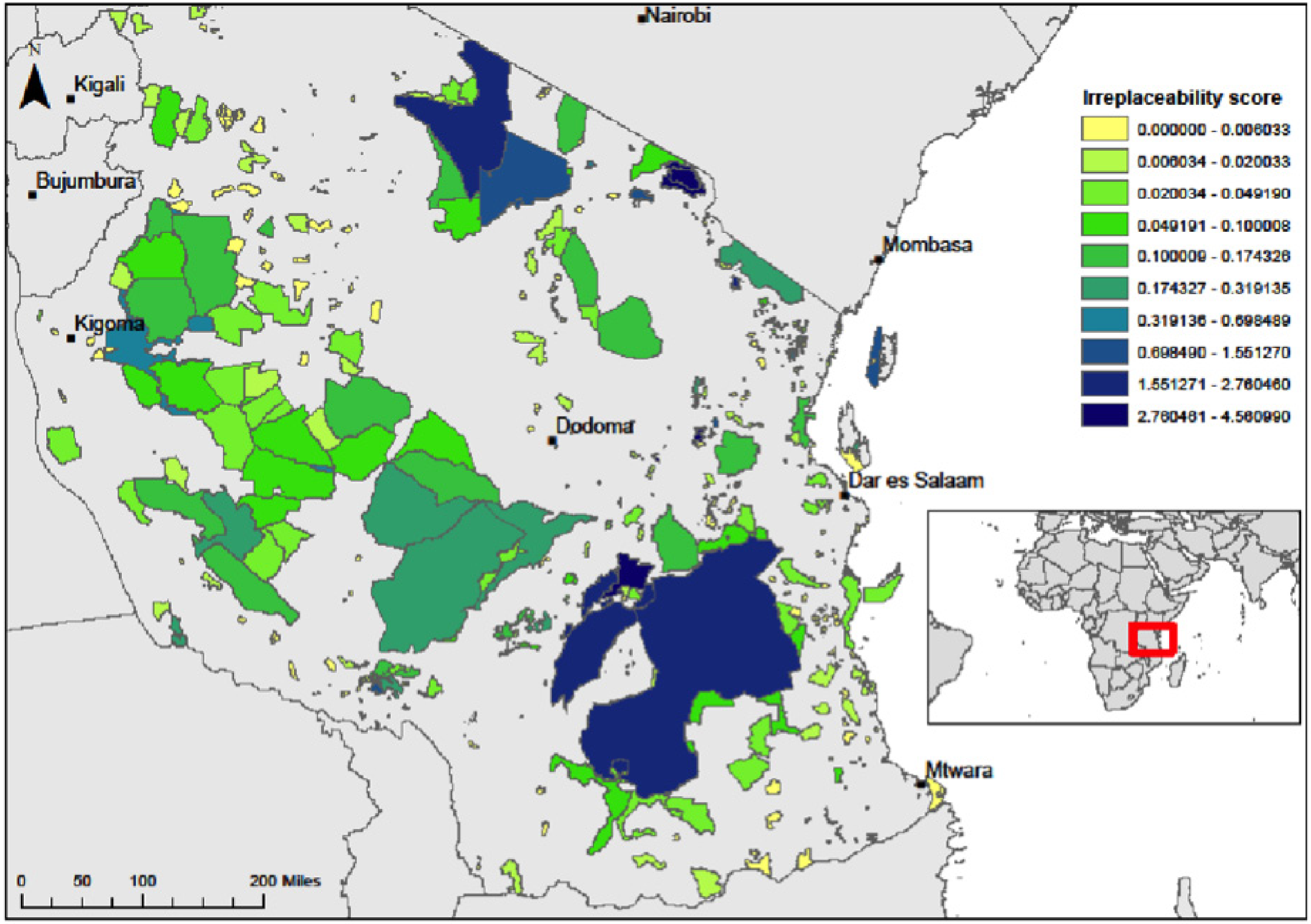


Figure 2.8: *Map of Tanzania's PAs, colour-coded by irreplaceability scores. Light yellow and green represent lower scores and darker greens and blues represent higher scores. Irreplaceability score is based on percentage overlap with bird, mammal and amphibian species distributions (taken from IUCN Red List of Threatened Species)*

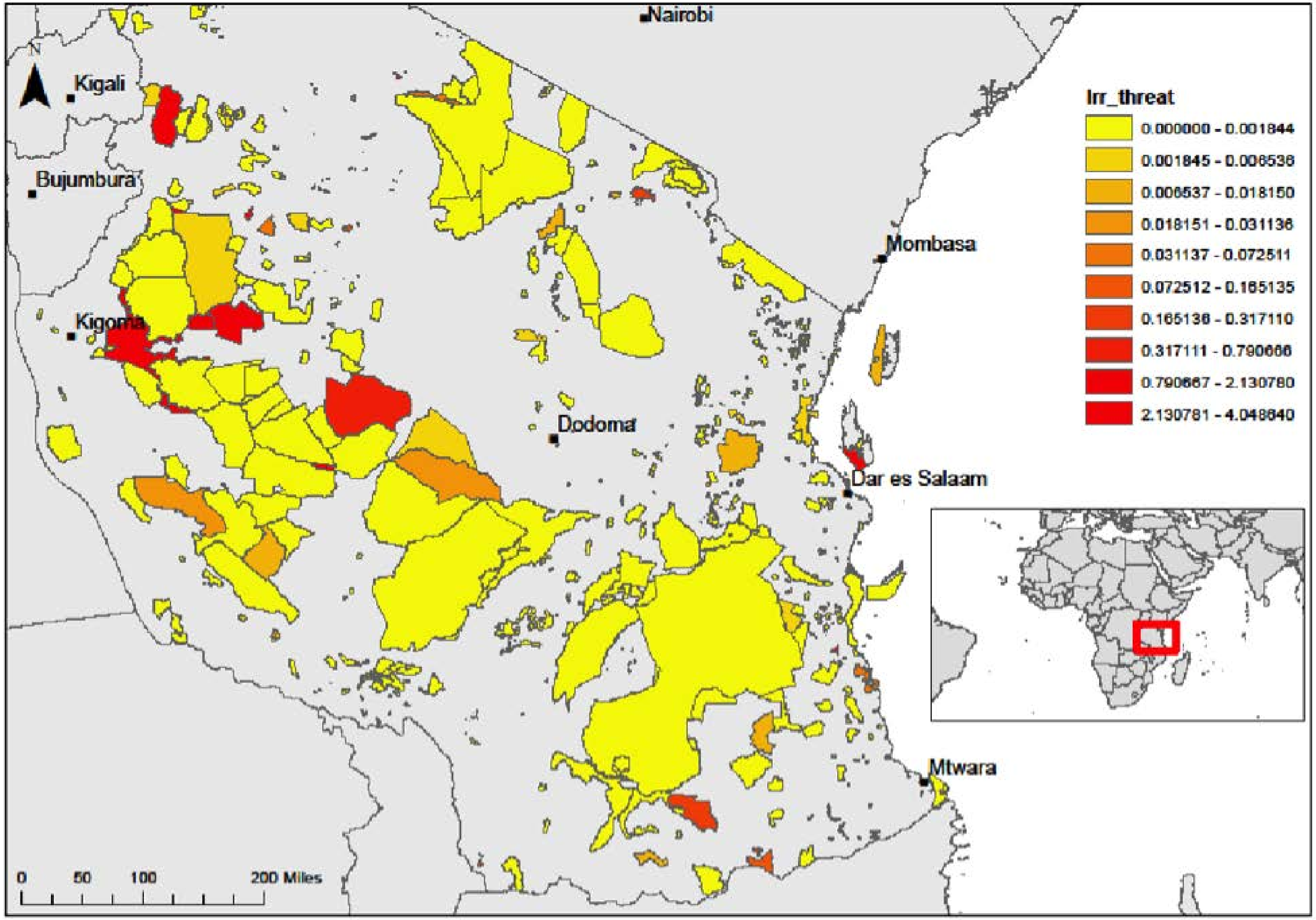


Figure 2.9: *Tanzania's PAs colour-coded by irreplaceability score based on threatened species of bird, mammal and amphibian. Yellows and light orange represent lower scores, while darker orange and reds represent high scores. Irreplaceability score is based on percentage overlap with bird, mammal and amphibian species distributions (taken from IUCN Red List of Threatened Species)*

2.8.4 Monetisable values

As well as the inherent value attributed to the PA network, there are also more tangible, monetisable values derived from present and future revenue generation for Tanzania's government and local communities. For example, taxes and fees for access to PAs such as national park entrance fees generate considerable revenue from the tourism industry (Table 2.6) and contribute a large percentage of GDP (Table 2.7).

International tourism in Tanzania, the majority of which is associated with PAs, increased by 41% between 2010 and 2013, but still remains relatively low in comparison to other sub-Saharan countries such as neighbouring Kenya, which received 1.4 million visitors in 2013, and South Africa with over nine million (World Bank 2013). Tanzania was ranked 11 out of 184 countries for relative importance of travel and tourism in long-term growth (2014-24). Summary information for the socioeconomic contribution of tourism in Tanzania, based on WTTC Annual Research country report (2015), is presented in Table 2.7.

Table 2.6: *Summary of the entrance and activity fees charged by the Tanzania National Parks Authority (TANAPA) at June 2015. All fees are for single entry for a period of 24 hours*

Fee type	Fee amount
Park entrance	\$30 – 100 (adults) / \$10 – 20 (children)
Entry for motor vehicles	TShs 20,000 – 150,000
Boat & aircraft	TShs 50,000 – 200,000
Accommodation	\$30 – 100 (person / night)
Tourist activities / guides	\$15 - 25
Filming	\$100 - 300
Crew	TShs 3,000 – 3,500

Additionally, Tanzania's wildlife represents an increasingly valued asset for sport and trophy hunting. The gross value of hunting tourism in Tanzania in 2008 was estimated at US\$25.3 million (Booth, 2010), for example, a 21-day lion hunt can cost up to US\$70,000 (Lindsey et al, 2012). However little of this value is captured by local communities or reinvested into PAs in which hunting occurs (Sachedina, 2008). Booth (2010), based on confidential financial records from private hunting companies, estimated that a hypothetical hunting company returns only 3% revenue to community development, and 23% to the Tanzanian government wildlife division, although this increases to 44% when indirect taxes and VAT are added. The remainder was captured by the private sector. The IUCN in a report on hunting concessions (2009) attributed only US\$0.04 per hectare in benefits gained from hunting by local communities.

***Table 2.7:** Summary data based on tourism report (2015): these cover all tourism but many travellers to Tanzania will be there specifically to experience the natural environment and wildlife within PAs*

Tourism value category	Monetary contribution	% Contribution to GDP
Direct contribution to economy	TZS 2,975.6bn	5.1% GDP
Total contribution to economy	TZS 8,252.7bn	14.0% GDP
Total contribution to employment	1,337,000 jobs	12.2% total employment
Visitor exports	TZS 3,365.3bn	21.9% total exports
Investment	TZS 1,864.5bn	9.5% total investment

2.8.5 Scientific research

PAs provide a valuable platform for scientific research, from ecology and sociology to archaeology and economic studies. Furthermore, international scientific research also brings in revenue for the government of Tanzania through travel by researchers and payment of fees associated with working in PAs (Table 2.8). Between 1985 and 2016, the National Science Foundation provided approximately US$11,794,640 to fund scientific research activities by American academic departments within Tanzania's PA network (NSF, 2015). Similarly, information on previous and current Natural Environment Research Council (UK) grants indicates funding of approximately £1,134,773 from 2010 onwards (Grants on the Web, 2015). A proportion of this money will be spent in Tanzania and thus can be considered a monetary value of research within PAs. The Tanzania Wildlife Research Institute (TAWIRI) coordinates all wildlife research conducted in Tanzania (including fees charged), funds research, provides training, and advises the government.

2.8.6 Cultural value and local communities

The plethora of natural beauty and wildlife across the Tanzanian PA network provides ecosystem services associated with human wellbeing, cultural, and aesthetic value. The majority of Tanzania's population is rural and pastoral, therefore any change in land tenure within and outside PAs affect a large proportion of Tanzania's population (Sendalo, 2009) and has implications for PA resilience. In recent decades, there have been government initiatives to involve local communities in management of the land and to promote benefits derived from PAs. One such programme is the Tanzania National Parks Authority (TANAPAs) Community Conservation Service (Leader-Williams et al, 1996), which includes benefits sharing through the Support for Community Initiated Projects (SCIP) Fund. This funding has been used for construction projects such as school and medical facilities, roads, and water projects. Between 2000 and 2008 approximately US$5.3 million was provided.

Although there has been a shift since the 1980s towards more community involvement in conservation and PAs, tangible benefits are often minimal, and revenue in many areas is not enough to offset the costs for local communities. Additionally, conservation initiatives involving local communities often emphasise economic incentives and ignore cultural values and traditions, which may be more important (Infield, 2001). The focus on community development also carries the risk of promoting land use activities that are harmful to conservation (Noss, 1997).

Table 2.8: *Summary of fees associated with conducting research within Tanzania's PA network set by TAWIRI (2012)*

Type of fee	Research type	Amount (US$)
Research within protected areas	MSc. And PhD students	51200 – 1800/year dependent on PA
	Visiting scientist	120 – 200/month
Key wildlife species research (Rhino, lion, cheetah, leopard, chimpanzee)	Foreign research project	2000/project/year
	Foreign-local collaborative project	1500/project/year
	Local project	100/project/year
Critical PAs (Serengeti, Gombe, Manyara, Kilimanjaro NP, Mahale NP, Ngorongoro conservation area	Forest project	2000/project/year
	Foreign-local project	1500/project/year
	Local project	100/project/year

Note: There are additional fees for the export of biological samples and specimens ($1-5 per sample / specimen).

2.8.7 Value capture

'High financial wildlife values captured by state agents, especially when coupled with high levels of corruption, create strong disincentives for central managers to devolve authority over wildlife to local communities' (Nelson and Agrawal, 2008).

Although there has been a drive to establish community-based management of PAs and natural resources in Wildlife Management Areas (WMAs), benefits and value from these are still captured by central actors via tourism and hunting for example (Nelson and Agrawal, 2008). Wildlife authorities continue to capture most of the value from international donors wishing to encourage community involvement, and without surrendering control over wildlife to communities.

Resources inside national parks are managed by TANAPA and thus value generated from national parks is captured by TANAPA. All wildlife and tourism outside national parks, which includes game and forest reserves and community land is under the Wildlife Division's jurisdiction. Commercial hunting is the Wildlife Division's primary source of income and, therefore, its management priority (Nelson and Agrawal, 2008). Little to no revenue from tourist activities and hunting is returned to local communities on these lands, and the rights over wildlife granted to the communities in the WMA framework are limited (Igoe and Croucher, 2007). Consequently, there has been a lack of progress in increasing the benefit from wildlife captured by local communities (Nelson et al, 2007).

Table 2.9: *Value capture summary for Tanzania PA network*

Value	Scale of value	Value capture
Tourism	National	Government Employees
Park fees for tourism	National – National Park / PA network	TANAPA (government) PA managers
Hunting concessions	Local – PA level value generation	Private hunting company Government – wildlife department Local communities
Tourism spending	Local and national	Government PAs
Research	National	Government (TAWIRI)
Ecosystem services	Local and National	local communities government Tanzanian population Wildlife Tourists / visitors

2.9 Risks

From the WWF Management Effectiveness Tracking Tool and Global Database on PA Management Effectiveness results for Tanzania, biological resource use and natural system modification are recorded most frequently by protected area managers when asked to score threat categories (Figure 2.10; WWF METT and GD-PAME, 2015). Within biological resource use there are four subcategories: 'Hunting, killing and collecting terrestrial animals'; 'Gathering terrestrial plants or plant products'; 'Logging and wood harvesting'; 'Fishing, killing and harvesting aquatic resources'. Within natural system modification there are six subcategories: 'Fire and fire suppression'; 'Dams, hydrological modification and water'; 'Increased fragmentation within protected area'; 'Isolation from other natural habitat'; 'Other "edge effects" on park values'; 'Loss of keystone species'. On average threats to PAs are scored between 0.8 (geological events) and 1.5 (biological resource use) (Figure 2.11).

Most PA managers in Tanzania scored the legal status of the PA highly, whereas PA regulations (e.g. to control hunting) and law enforcement were scored lower (i.e. were perceived as less adequate), and protection systems for controlling access and resource use was mostly scored either 1 or 2 (i.e. of moderate effectiveness) (Figure 2.12).

Figure 2.10: Frequency for which a given WWF-METT threat category was assigned a score (i.e. 1 to 3) for Tanzania's PAs that have a completed the questionnaire (N=215). For each category there are between two and seven subcategories

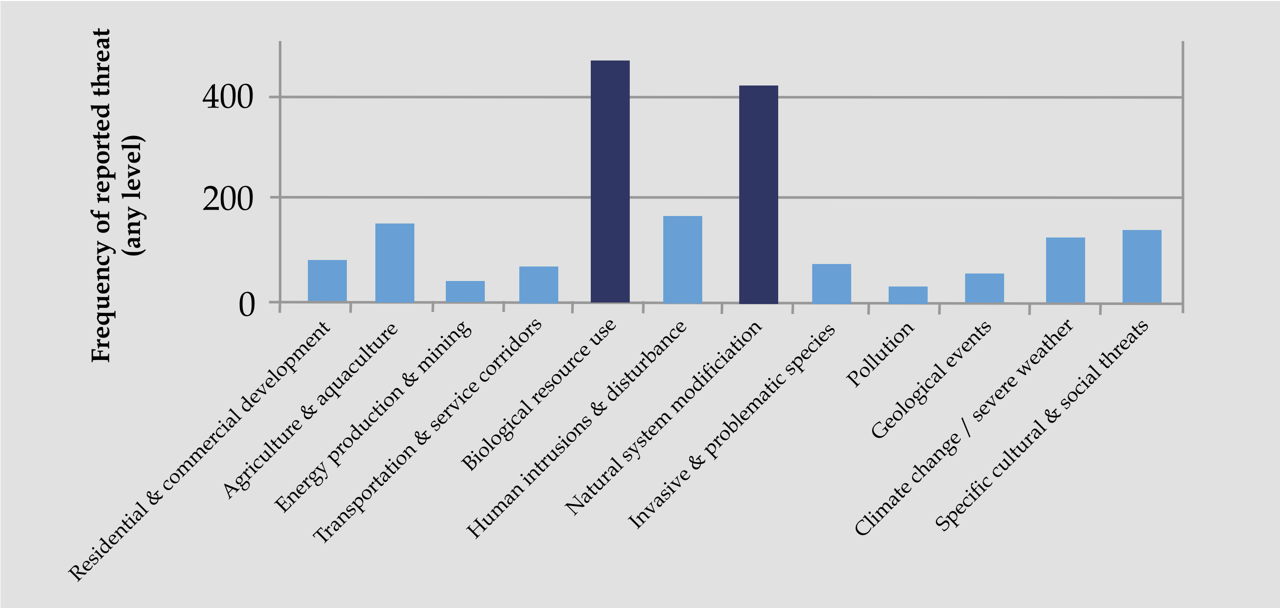


Figure 2.11: The average score allocated to the 12 WWF-METT threat categories across the PAs in Tanzania that have a completed questionnaire. Scores for each threat ranged from 0 to 3 with 3 being the highest level of perceived threat. NAs were not counted as zero scores

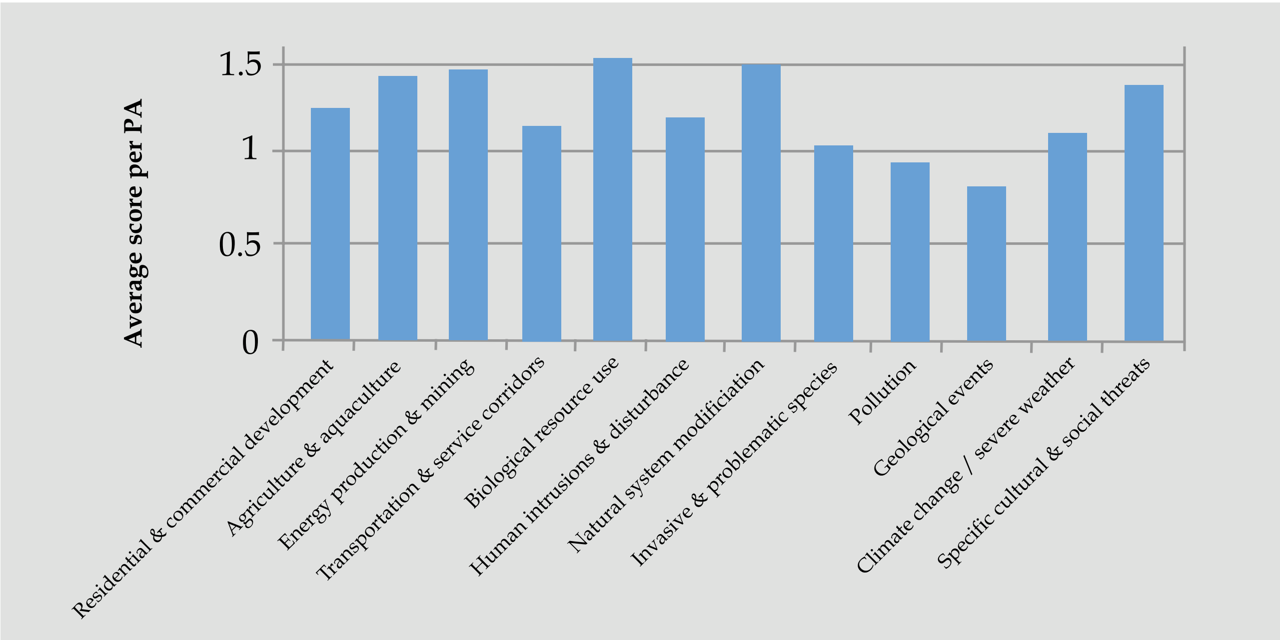


Figure 2.12: *Scoring for the additional threats to PAs in Tanzania: the legal status of the PA, whether PA regulations and law enforcement are in place, and whether there are systems to control access and resource use*

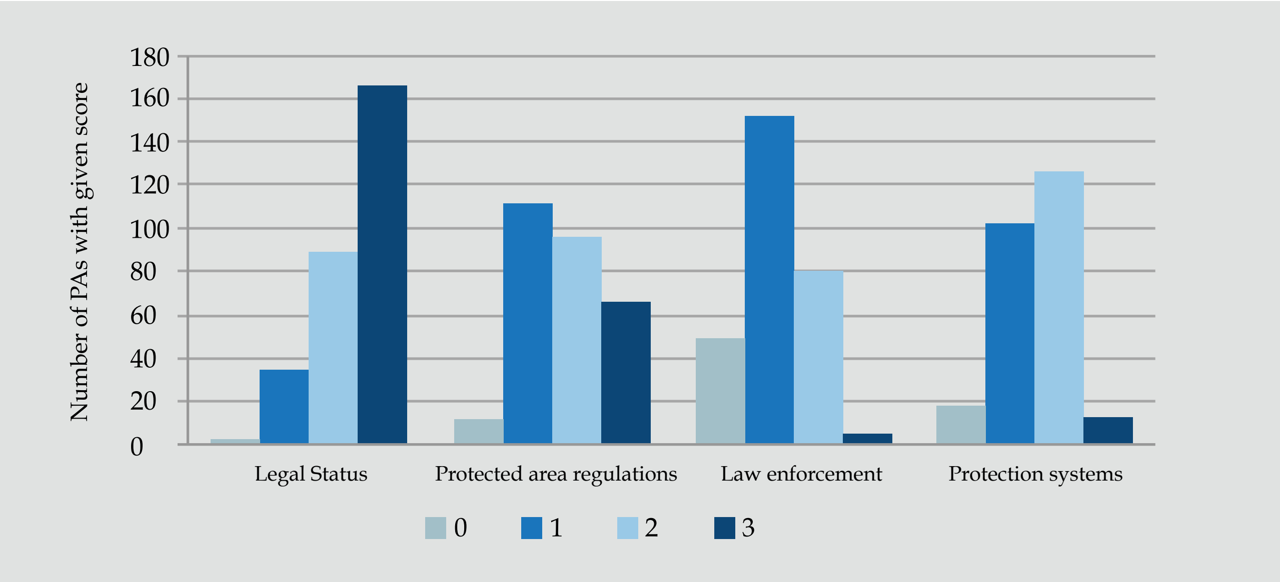


Table 2.10: Information is provided on a selection of current and future threats facing the PA network of Tanzania with specific examples given where possible. A status is given to the threat depending on seriousness: green is minor, orange moderate and red severe. The timescale of the threat is also indicated.

Threat	Overview / examples	Status / timescale
1. Physical climate change	Climate change, including increased temperatures in sub-Saharan Africa may lead to range shifts of many species in both latitude and altitude. Some protected areas will gain new complements of species and others will lose certain species (Hole et al, 2009), therefore there is a need to ensure the PA network as a whole can accommodate these shifts. Climate change is also likely to lead to further loss of habitat as temperatures rise and extreme weather events increase.	Orange Long and short-term
2. Illegal wildlife trade and poaching	As with the majority of protected areas across sub-Saharan Africa poaching and illegal bush meat hunting pose a great threat to the value and persistence of Tanzania's protected areas. For example Selous Game Reserve has been on the UNESCO list of World Heritage Sites in danger since 1986 due to illegal activities; and since 2011 due to reduction in elephant populations; and in 2015 due to significant reduction in animal populations (elephant and rhino) from poaching (UNESCO, 2015).	Red Long-term and increasing in frequency and intensity
3. Land encroachment: Deforestation	Tanzania is a country where otherwise unmet demands for food and fuel lead to massive forest degradation and deforestation (Fisher et al, 2011). The main causes are overgrazing and unsustainable pole-cutting, logging and firewood collection as well as man-made wildfires. According to Global Forest Watch there was a net loss of 1,173,554ha between 2001 and 2013. Tanzania had the sixth largest annual net loss in forest area between 2000 and 2005 in the world of about 412,000 ha/yr (www.unep.net).	Orange Continuous and Long-term impact

Table 2.10: *Continued from previous page*

Threat	Overview / examples	Status / timescale
4. Extractive industries	Although the mining industry contributes half the percentage to GDP that tourism does (~3%, Tanzania Ministry of Finance, 2014) it is a growing industry, particularly for minerals, and threatens PAs and the wider environment with deforestation, habitat degradation, and pollution. Due to the large area of Tanzania already protected, conflict over land use due to the expansion of mining operations is foreseeable in the future. Example 1: Uranium mine on the edge of Selous Game Reserve threatens the habitat and wildlife with radioactive waste seeping into the reserve in surface water (Friends of Serengeti, 2015). Example 2: Soda ash factories are proposed on the shore of Lake Natron, threatening the ecological integrity of the lake and nesting site of an important population of lesser flamingo (Phoenicopterus minor). This is despite an analysis show that soda mining is not economically viable and that investing in tourism and protection of the environment returns greater gains, particularly for local communities (Birdlife International, 2012). Example 3: Illegal gold mining in the forests of the Eastern Arc Mountains, leading to forest clearing and degradation from mining activities and miner's camps. Since its peak in 2004 this threat has been reduced (Burgess et al., 2005). Attempts have been made to minimise the risk to the environment from mining operations, including collaboration between the Tanzania Chamber of Minerals and Energy (TCME) and WWF, which provided a guidance document for incorporating environmental considerations into policy and project formulation. This document was aimed at potential investors and government officials responsible for assessing such investment (WWF and TCME, 2012).	Red Short term impacts from land-clearing during construction. Longer term impacts from pollution, infrastructure, lost tourism
5. Unsuitable infrastructure development	Example: Commercial road through the northern region of Serengeti National Park was proposed in 2010 as a means to link the coast of Tanzania to Lake Victoria and surrounding countries, something which has been under discussion for several decades. Dobson et al. (2010) voiced concern over the impact of the proposed road on one of the greatest animal migrations on the planet – the wildebeest migration. Simulations suggest a population decline to 300,000 due to blocked migration would increase forest fires, diminish grazing quality and release CO_2 (Gereata et al. 2009). Others disagree and say opposition based on a single factor is misleading and that the road offers potential benefits for the Serengeti ecosystem (Fyumagwa et al., 2013). In 2014 the proposal was declared unlawful in a case of Tanzania government versus the African Network for Animal Welfare.	Orange Short-term negative impacts from destruction and degradation of habitat Long-term negative impacts from habitat degradation, pollution Long-term positive impacts could include better access to certain PAs

***Table 2.10:** Continued from previous page*

Threat	Overview / examples	Status / timescale
6. Regulatory change: Protected Area Downgrading, Downsizing, and Degazettement (PADDD)	Instances of PADDD of protected areas are widespread globally (Mascia et al., 2014) Most PADDD events represent trade-offs between policy for biodiversity conservation and other policy, for example to do with infrastructure or development. The online resource that tracks PADDD of protected areas around the world PADDDtracker.org identified 27 PADDD events across the Tanzania protected area network, which are summarised in Table 3. Moreover, PAs in Tanzania have been degazetted without following legal procedures. Some PAs degradation is so high through human activities that there has been de facto degazettement and no restorative action from government.	Orange Long-term: This is a real and continued risk and undermines management efficiency
6. Regulatory change: Lack of IUCN management criteria	Lack of allocated IUCN management criteria for almost 80% of Tanzania's protected areas (Protected Planet, 2015).	Green Becomes more important when other threats are present as management criteria determines strength of resources for maintaining PA integrity.
7. Conflict between communities and authorities: Poverty Land tenure	Tanzania is one of the poorest countries in the world with an estimated 28% of the population living below the poverty line in 2012 (World Bank). Rural communities in particular are highly dependent on natural resources, and affected by any restrictions on access to these resources. Pressure to add more land under protection, which is reflected in recent policy. Local communities are more inclusive in protected area management than historically but conflict over the establishment of reserves still arises and threatens the natural state of protected areas.	Orange Long-term.
8. Corruption	Example: in 2006 it was reported that approximately half the money Norway spent on a Management of Natural Resources Programme was lost to corruption and mismanagement by the Ministry of Natural Resources and Tourism. Furthermore, in 2012 another case of misuse of REDD funding was reported (red-monitor, 2013). In a study where national-level decision makers were questioned about threats to, and management of, PAs, reduction in corruption was valued as the second most important in ensuring effective management (Stellmacher et al. 2012). Indeed some decision makers estimate that only 50-60% of funds allocated for development projects in Tanzania go towards intended use.	Green Intermittent threat, difficult to predict.

Table 2.10: *Continued from previous page*

Threat	Overview / examples	Status / timescale
9. Local conflicts and geopolitics	Increasing land scarcity is leading to the spread of land users to marginal lands where resources are more limited, such as semi-arid land, resulting in conflict between land users and interest groups (Sendalo 2009). Conflict over water resources, e.g. in Pangani River Basin – experiencing influx of pastoralists (Inter Press Service, 2013). The threat to PAs depends on the strength of enforcement and management in maintaining the integrity of borders.	Green Short-term, likely to increase
10. Breached ecological thresholds	With a larger proportion of the rural population living as pastoralists, grazing intensity can lead to shifts in the stability and composition of plant communities. Similarly reductions in grazing intensity by wild ungulates, for example following drought and associated population decline, can lead to shifts from grassland to woodland (Dublin et al. 1990).	Green Long-term, ongoing
11. Shifts in visitor preferences	Activities that degrade protected areas not only threatened their continued effectiveness at conserving nature but also have an indirect effect on lowering their appeal as wilderness areas for tourists and subsequently affect employees of the tourism industry, many of whom are local residents. A little considered threat to the Tanzania PA network is the tourism bias in use of the northern circuit that includes the Serengeti NP and Kilimanjaro over the southern circuit that includes Ruaha NP and Selous. This bias has the potential to lead to a lack of investment and resources in the southern circuit PAs, for example in infrastructure and management (GEF, 2009).	Green Long-term, potentially increased threat in the future

2.10 Investment

2.10.1 Historical investment

The Wildlife Division of the Ministry of Natural Resources, which deals with wildlife policy and regulation, relies on government funding, whereas TANAPA, which manages Tanzania's national parks, and TAWIRI largely operate independently of government, receiving donor funding.

After independence in 1961, the potential income generated by Tanzania's wildlife meant tourism became a national priority as a means to support economic development; the government reversed its stance on conservation and made additions to the PA network. There has been a convergence of private investors, development agencies, international conservation NGOs in agendas and investment targeting (Levine, 2002), which can be best understood in the context of international development policy and association with conservation NGOs.

Between 1981 and 1985 Tanzania implemented economic reforms and foreign companies were allowed to control business and tourist companies. In 1981, WWF established a discretionary fund for anti-poaching and to assist TANAPA. WWF also helps fund the Community Conservation Service (CCS), a programme to integrate local communities into management of PAs. The National Economic Empowerment Policy of 2004 created a favourable environment for investment and economic growth (with the focus on tourism). It was also intended to give Tanzanians easier access loans to boost internal tourism.

PA networks in Africa have a long history of underfunding, and for Tanzania it has been estimated that there is a US$2,465,681 annual shortfall in funding for the PA network (Hanks and Attwell, 2003). Often the shortfalls in PA budgeting for natural resource/wildlife sector are due to poor presentations of the value of the sector. Funding and investment in Tanzania's PA network comes from a variety of sources (Table 2.11). World Bank operations in Tanzania suggest a total investment of $346.2 million since 2001 for improving management of the environment and ecosystem services (World Bank, 2015). Investment often comes from several donors including international and national sources. Other than external sources of investment, PAs generate revenue from visitors and other activities, which can then be reinvested. For example, review of TANAPA's Quick Reference Statistics (1987-2001), which summarise budgets and revenues of the national parks, shows four national parks (Kilimanjaro, Serengeti, Lake Manyara and Tarangire) generated enough revenue to support operations in the remaining parks (Hanks and Atwell, 2003).

Table 2.11: *Examples of investment in Tanzania's PA network*

Investor	Type	Examples of investment
Wildlife Conservation Society	NGO	Research and monitoring systems. Protected area design and management, community conservation
WWF	NGO	Develop integrated water management approaches. Sustainable use of natural resources. Environmental awareness creation. Capacity building for biodiversity conservation. Policy development, implementation and harmonization

***Table 2.11:** Continued from previous page*

Investor	Type	Examples of investment
Water Aid	NGO	Pumps installation Water functionality and distribution of water points Education and awareness on sanitation issues
Conservation International	NGO	Critical Ecosystem Partnership Fund from 2003 (CEPF) – US$7 million in conserving Eastern Arc mountains and coastal forests of Kenya and Tanzania (in collaboration with GEF, World Bank, Government of Japan, and John D. and Catherine T. MacArthur Foundation.
Department for International Development (DFID)	Bilateral Development Cooperation Agency	To promote sustainable management of Usangu Wetlands Catchment. Support livelihoods programmes and Wildlife Management Areas (WMA)
USAID	Bilateral Development Cooperation Agency	Support other NGO like WCS to implement some of its objectives. Support capacity building programmes to the WMAs
African Development Bank	Multilateral Banks	
Global Environment Facility (GEF)	UN-administered funder	Strengthening the Protected Area Network in Southern Tanzania: Improving the Effectiveness of National Parks in Addressing Threats to Biodiversity (GEF, 2011) – US$17, 364,500. There is a funding bias towards northern circuit which includes the Serengeti and Kilimajaro Address PA management Increase connectivity between PAs, better management capacity and financing Provide infrastructure and field equipment Critical Ecosystem Partnership Fund from 2003 (CEPF) – see Conservation International
UNEP Finance Initiative	UN-private sector partnersip	
GTZ	German Development Agency	Selous Game Reserve
The African Wildlife Foundation		Arusha, Serengeti, Tarangire and Lake Manyara National Parks
CARE International	Humanitarian organisation	Conservation programs in the Jozani and Ngezi Forest Preserves and Eastern Arc Forests.
Friedkin Conservation Fund of Tanzania	Charitable Trust	Work closely with the Wildlife Division of Tanzania. Anti-poaching operations, innovative community development projects, and Geographic Information System (GIS) mapping
Tanzania Forest Fund	In-country NGO	Sustainable management and conservation of forests

Source: Hanks and Atwell, 2003; Levine, 2002

2.10.2 Future investment

The Tanzanian government is encouraging tourism from emerging markets in China, India and Russia. For example, the northern circuit for tourism has reached capacity, therefore, China has been targeted for potential investment in developing infrastructure and access across the southern circuit of PAs, particularly Selous Game Reserve and Ruaha National Park, both of which are impassable during the rainy season (IPP Media, 2014). Tanzania has previously sought investment in the tourism market from Chinese investors (All Africa, 2013).

2.10.3 Management practice

A global study of PA management effectiveness, based on 4,151 assessments, concluded that 27% of PAs show major deficiencies in management, and 13% are considered 'paper parks' with no management effectiveness (Leverington et al, 2010).
A World Conservation Monitoring Centre analysis found that reporting on capital investment was patchy for most countries and that the PA budget for Tanzania was considerably below the global mean (James and Paine, 1999).

- Tanzania (1994): based on 41,131km2 had a budget of US$182 per km2, US$47 capital investment, US$19 foreign investment with a shortfall of US$160.
- Staffing: 1,400 total (1,298 field, 35/1000km2), which is lower than South Africa, Zimbabwe, Malawi, or Kenya.
- Budgets correlate more highly with per capita income than staffing, suggesting some developing countries, such as Tanzania employ substantial staff on a lower budget, and conservation efforts should not be judged solely on budget/income/investment.

Based on WWF METT data: Average budget (US$) = 11,945,409 (with highest budget of 1,141,420,000 for Saadani, TANAPA managed; figures correct as of 2014)

- Only 54 PAs have actual figure for their budget from the METT.
- Most PAs score poorly on the three budget categories (Figure 2.13).
- 76% of respondents to the METT gave low scores to budget adequacy, security and management (i.e. total < 5; Figure 2.14)
- Management of the budget most frequently scored 0; adequacy and security of the budget most frequently scored 1.

Figure 2.13: Summary of protected area managers' assessment scores for the adequacy, security and management of their budget. Bars represent the number of PAs per score class, split by the three questions regarding budget: 1. 'Is the current budget sufficient?' (Adequacy of current budget); 2. 'Is the budget secure?' (Security of budget); and 3. 'Is the budget managed to meet critical management needs?' (Management of budget). Scores rank from 0 (poor) to 3 (excellent). Where a PA had a low score in one category it often had a low score in all three (or the same score for each)

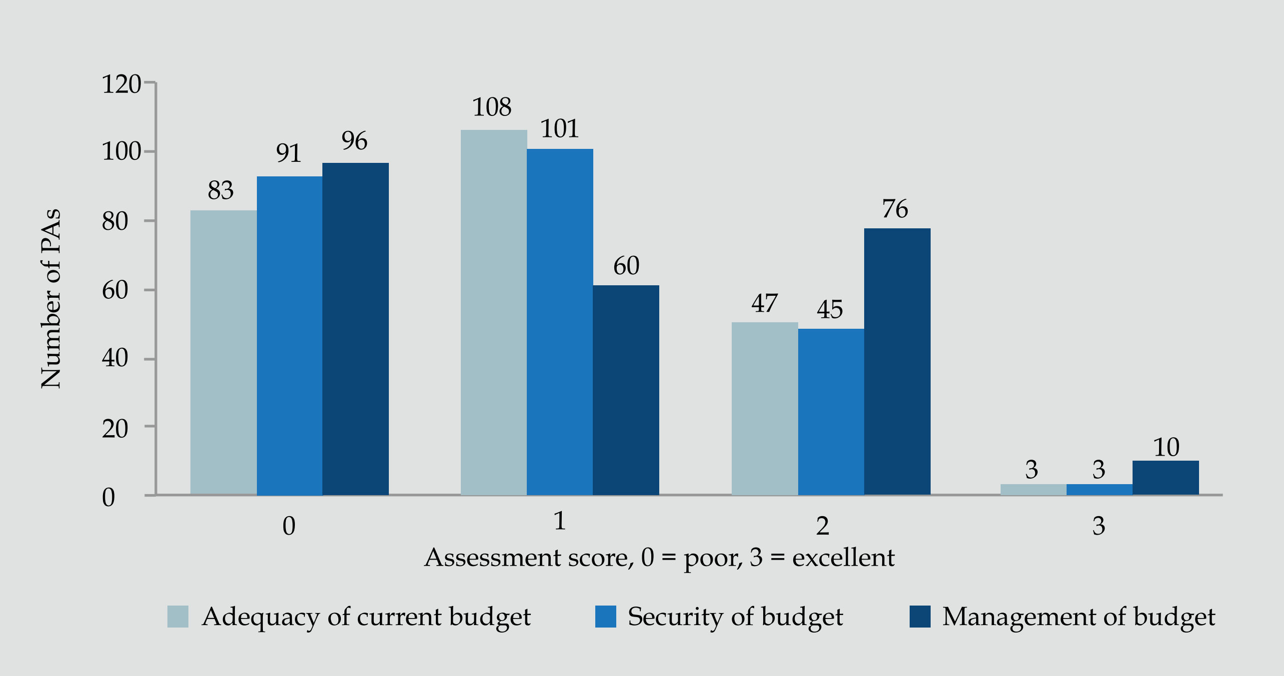
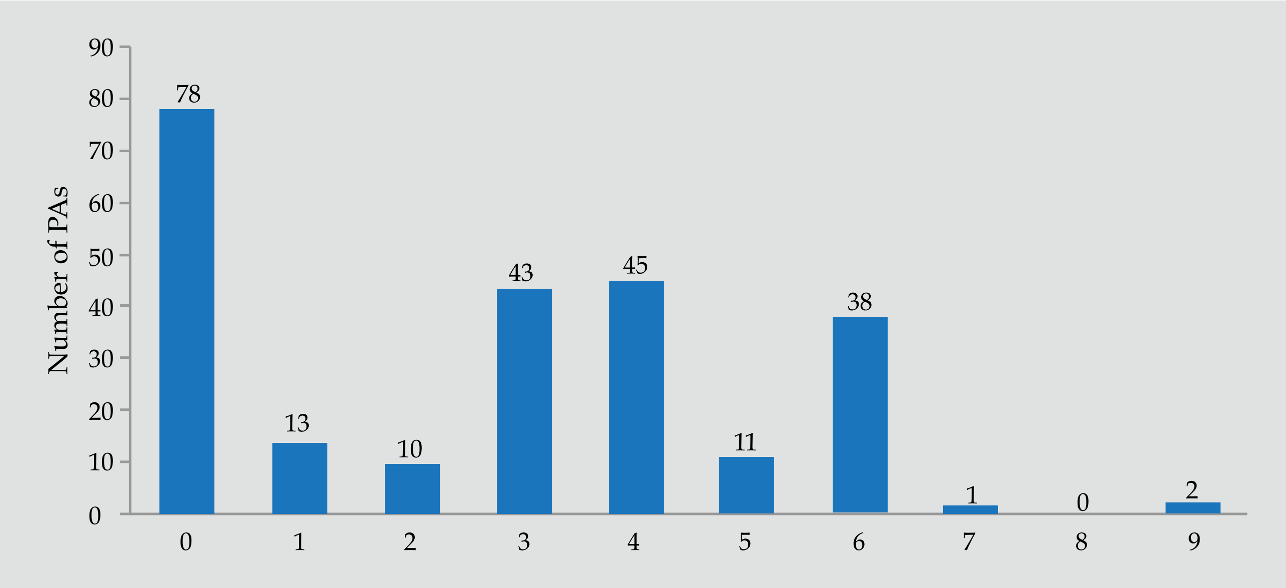


Figure 2.14: Summary of the number of PAs with a given total score regarding the three budget categories in the WWF Management Effectiveness Tracking Tool. Minimum = 0; maximum = 9



PART II: Supplementary Materials

Part II of this report contains more of the underlying research, literature, and data that helped to underpin Part I. This has been separated out to ensure that the main conclusions in Part I could be presented in a concise manner, particularly as the intended audiences for this report are broad – from conservation practitioners to policymakers.

This part of the report provides researchers with additional information and signposting that could usefully support further research in this and related areas. Part II also presents more detail on the approaches we used and literature we systematically reviewed. Here we also offer a richer snapshot of the state of values that PAs generate, the investments being made into PAs, and the current and emerging threats they face.

Section 1: Values/benefits literature review and meta-analysis

Authors: Ben Caldecott, Harriet Milligan, and Dexiang Chen

In Part I we outlined how PAs consist of five types of asset: i) biophysical assets, ii) human assets, iii) infrastructure assets, iv) institutional assets, and v) cultural assets. These assets generate monetisable and non-monetisable values that are captured by a diverse range of beneficiaries or groups in society. Figure 1.1 gives an overview of how these different groups benefit from the forms of value created by protected areas. For example, nations and polities gain a sense of identity from PA networks, and enjoy public health and environmental benefits arising from the natural systems that PAs maintain; economies benefit directly from revenue generated from tourism, natural resources and ecosystem services; organisations gain value from developing networks within and between nations, funding flows and prestige; and finally at the individual level, citizens capture cultural and intellectual values and wellbeing. In subsequent sections we examine who captures this value, how they capture it, and how an understanding of value capture can help inform PA policy and management. Here we examine the types of monetisable and non-monetisable value generated by PA assets.

Figure 1.1: *Value creation*

1.1 What are values?

Value relates to the worth of a certain object or the benefit of a particular activity (Perlman and Adelson, 2009). It is the actual or perceived positive qualities seen in things or as a result of actions (Barry and Oelschlaeger, 1996). Value is often measurable using either direct or indirect methods.

Competing uses of the finite natural environment, for farming, development or as a natural space, requires society to make the best choices (Phillips and Costa, 2005). The benefits from farming or development are obvious as we are familiar with their monetary value captured in markets (Phillips and Costa, 2005). However, many forms of value generated by PAs and by nature do not appear in markets (Kallis et al, 2009). It is important to highlight that even though these forms of value are not directly monetisable, they are nonetheless critically important (Gomez-Baggethun and Ruiz-Perez, 2011).

When resources are scarce, there is a need to make allocation choices and monetary value has been traditionally used to provide a basis for such decisions (Constanza, 2001). Unless assets are assessed to reflect the values they generate, conservation of non-valued or undervalued assets could be hindered, as they could be mistaken for inferior choices relative to those that have been given explicit monetary value (Gomez-Baggethun and Ruiz-Perez, 2011).

1.2 Valuing nature

In hopes of raising awareness for biodiversity conservation, natural resource scientists, such as Ehrlich and Ehrlich (1981), Helliwell (1969), Odum (1972) and Westman (1977), explored concepts of nature's services that stress the benefits ecosystems can generate for society. This perspective demonstrates the wide array of values that nature can provide to people, broadening the range of motivations for conservation (Perrings, 2009). In addition, an emphasis on the essential services provided by nature as prerequisites for long-term economic sustainability instead of 'conservation versus development' (Folke, 2006) bridges the gap between the worlds of natural science, social science and economics, resource management and development, and public and private policy, helping to transform polarised critiques into constructive discourse (Braat and de Groot, 2012).

1.2.1 Development of the value of nature

Rapid industrial growth, technological development, and capital accumulation in the 19th century sparked a paradigm shift in the valuation of nature. Towards the end of the 19th century, huge changes in landscape and unprecedented extirpation of flora and fauna sparked off a huge social movement in the West, rapidly promoting a radical change in perception towards the relationship between nature and people (Jepson & Whittaker, 2002). Since then, a plethora of approaches has been developed at various scales to achieve conservation goals (Ladle & Whitaker, 2011). However, despite more than a century's worth of conservation efforts emphasising the value of nature and the importance of curbing the decline in biodiversity, the losses remain unabated (Armsworth et al, 2007; Butchard et al, 2010).

1.2.2 Novel techniques to value nature

The latter half of the 20th century saw reinvigorated interest in valuing nature, brought about by the wave of modern environmentalism (Gomez-Baggethun and Ruiz-Perez, 2011). Non-marketed ecosystem services are viewed as positive externalities that, if valued in monetary terms, can be more explicitly incorporated in economic decision-making. Previously unaddressed values have been identified and a comprehensive picture of the value of nature has begun to emerge (Gomez-Baggethun and Ruiz-Perez, 2011).

Subsequently a range of monetary valuation techniques has been developed and increasingly refined to identify these different value types (Braat and de Groot, 2012). The techniques typically rely on related marketed goods and services as proxies, or on observed consumer behaviour (Gomez-Baggethun et al, 2010). Valuation studies have also begun to develop tools that accord value to expected consumer behaviour in hypothetical markets simulated through surveys.

1.2.3 Protecting nature via the articulation of values

Despite critiques that cautioned against the use of valuation (Kallis et al, 2009; Gomez-Baggethun and Ruiz-Perez, 2011), it is necessary to ensure that nature is not taken for granted and undervalued (Foster, 2002). The developments in the valuation of nature supported by advances in techniques over recent years has afforded new conservation strategies, such as the calculation of natural capital. Incorporated into national accounting, natural capital initiatives can enable the regulation and monitoring of nature values (Azqueta, and Sotelsek, 2007).

1.3 Protected Area Values: a review of the academic literature

1.3.2.1 Data acquisition

We conducted a systematic review of the literature for the time period 1977 to 2015 by searching the Scopus online database. The search terms used to generate a list of potential studies for inclusion in the review were: mone* OR econom* OR valu* OR benefit*) AND (protected area* OR national park* OR reserve* OR private park* OR conservation area* OR biodiversity OR nature) in the title, abstract or keywords of the article.

Considering the variety of approaches values were grouped in different studies, we have decided to adapt a streamlined version suggested by Pabon-Zamora et al (2008) as they are the most up-to-date value groupings used in valuing nature in protected areas. The five value grouping considered are 1) economic, 2) health and spiritual, 3) intellectual, 4) recreational, and 5) resources and services (see Box 1.2 below for detailed definitions on each value grouping). For each study identified as relevant to the mentioned value groupings, the following data was extracted: source, year, publication type, geographical scale, country, specificity of valuation, specificity of asset, asset type, degree of focus on monetary value, actual monetary value, methodology (see Box 1.1), beneficiaries, funding body.

1.3.2.2 Resultant data set, assumptions and limitations

The search returned approximately 5,200 articles, which were then considered for inclusion based on a scan of the abstract and results, with a focus on selecting those with monetary value and based on a particular protected area or protected area network. The final dataset contains 101 studies from 47 journals covering five main geographic regions relatively equally (Figure 1.2 below).

The systematic review focused on 12 different value types, however attempts to analyse each of the 12 values individually revealed uneven distribution, inconsistent use and high overlap due to ambiguity in the definition of values. In order to reduce double counting we have combined the values into five distinct groups (see Box 1.2). These groups are (components in brackets): intellectual value (scientific and learning); health and spiritual value (intrinsic, spiritual, therapeutic, and cultural); resources and services value (biodiversity and life sustaining); economic value (economic and future); recreation value (recreation) and are discussed further with reference to temporal and geographic trends in the literature below.

In addition, the analysis also contains the following assumptions and limitations:

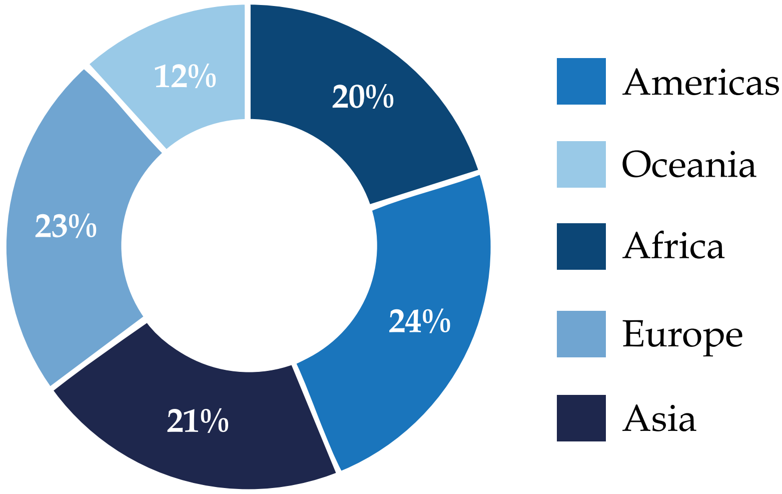
Global studies: global studies are assumed to have considered all regions equally. Hence, all global studies encountered are assessed in the count for all five regions in our analysis. Thus there is potentially an overestimation of the number of studies for each region.

Timing: many studies did not mention the exact timing or period of study they were referring to. We assume that, unless specified by the author, all values and valuation refer to the year in which the article was published.

Box 1.1: *Valuation methods*

Different methodological approaches were used to estimate value depending on the context and aim of a given study. For example, those studies with a focus on benefits derived by tourists often used contingent valuation methods (willingness-to-pay).
Social or stated preference valuation methods include: contingent valuation method (willingness-to-pay), which involves asking individuals to give a specific monetary value to a protected area or ecosystem service. These methods can be used to measure both use and non-use benefits (Jones-Walters and Mulder, 2009), however they rely on the perceptions of the public, who may be ill-informed.
Revealed preference methods are based on identifying the best option based on stakeholder behaviours, and include hedonic pricing, travel cost method, production function analysis, prevention cost method, and averting behaviour method.
Total Economic Value (TEV) is a monetary value method that attempts to include all expressions of value of a protected area or ecosystem, including both use and non-use values. It is an aggregation of all the functions and services of a given ecosystem, and equates to the total benefits expressed in monetary terms to be used in cost/benefit analyses. However ecosystems are not simply the sum of the different parts and it is difficult to calculate the overall value of an ecosystem (TEEB, 2008; Jones-Walters and Mulder, 2009).

Figure 1.2: *Geographical distribution of studies included in the final dataset*



Missing full-texts: in addition to journal articles, the Scopus search captured book chapters, proceeding reports and various forms of publication with abstracts that suggest relevance. In addition, there were also peer-reviewed articles, which do not have full-texts available in Scopus. In order to be consistent, we only selected articles from peer-reviewed scientific journals with full-text available in Scopus. Hence, this bias probably underestimates the number of peer-reviewed publications in our study.

Specificity: we considered all literature that mentioned or discussed protected area values regardless of their specificity. However, there is a disparity, which ranged from studies with low focus on values, typically mentioning them in remarks, diagrams or tables, to articles that focus solely on a specific value.

Methodology: economic values reported in the studies were obtained by different methods. Although different methodological approaches come with their typical limitations and assumptions, these are not considered in our analysis and values are presented as expressed in the studies.

The method used to assess value within a given study, and the resulting estimates of value for PAs and the services provided by such natural areas is partly dependent on the authors' interpretation of value. The main splits are summarised below as intrinsic versus extrinsic; utilitarian versus non-utilitarian; marketable versus non-marketable. Typically, it is easier to assign economic value to utilitarian (use) values and marketable values. The values considered in the literature review here come under these groupings but not always exclusively to one or the other, for example biological diversity could be classed as an intrinsic, non-utilitarian value of a PA, something that adds value simply by its existence. However, as shown by assessments of ecosystems services, biological diversity is frequently integral in ecosystem functioning and subsequently for many provisioning and regulatory services (Hooper et al, 2005).

Intrinsic versus extrinsic values

The intrinsic value of nature is that all human and non-human life (and abiotic features) have a right to exist regardless of benefit or cost to humanity. Biodiversity results from billions of years of evolution and should continue. The intrinsic value is not placed on nature due to human perceptions of its value and is irrespective of whether we use or see it. Conversely, extrinsic value depends on the relationship of biodiversity or a protected area with something else, most often people; it is context-dependent.

Box 1.2: *Definition for value groupings adapted from Pabon-Zamora et al (2008)*

Value group 1 – Intellectual value

This group is comprised of scientific values and learning values; they are grouped together since the distinction between scientific endeavours and individual learning experience for a visitor in protected areas is often unclear. Aesthetic value is also included in this group representing intellectual value generated by intrinsic beauty, such as through art and literature.

Value group 2 – Health and spiritual value

This group is comprised of intrinsic, spiritual, therapeutic, and cultural values. The intrinsic value of a PA is associated with the belief that nature has a value separate from humanity and humanity's need for it. PAs have value in maintaining cultural diversity, particularly of those local communities integrated with PAs. Culture incorporates a society's way of living, traditions, values and crafts. Spiritual values relate to the meaning and value a particular group of people (culture, local community, religion) attribute to nature or a PA, or the spiritual meaning and sense of wellbeing resulting from a visit to a PA by an individual.

Value group 3 – Resources and services value

This group is comprised of biodiversity and life-sustaining values from which a multitude of direct and indirect resources are derived locally, nationally, and internationally. Biodiversity value is also realised through the protection of diversity including species, genetic, and functional diversity. Additionally life-sustaining values are those linked with the services provided by PAs such as regulatory services (water and nutrient cycling, climate and weather), provisioning services (food, fuel, products), and mitigation or insurance against future events like the effects of climate change (flooding, storm damage). This group also includes future value, which is ascribed to a PA based on the knowledge it will provide benefits in the future.

Value group 4 – Economic value

This value category encompasses the majority of the other listed value categories to differing degrees due to their potential economic return (current or future). PAs are often important local and national revenue-generating assets with adequate investment. Revenue comes in the form of resources (e.g. food, medicine), tourism, and provisioning and regulatory ecosystem services (e.g. water purification, carbon sequestration, flood control).

Value group 5 – Recreation value

Recreation value is synonymous with the value of PAs for tourism. Tourism is possibly the greatest revenue generator for PAs globally and a recent study estimates that PAs receive around eight billion visits a year generating global gross expenditure of ~US$600 billion/yr (Balmford et al, 2015). This indicates that a huge number of people place value on PAs for recreation and pleasure. They visit PAs for various reasons: ecotourism such as safaris, diving, tours, walking and other outdoor pursuits, or to gain wellbeing.

Utilitarian versus non-utilitarian (use and non-use) values
This distinction is related to the theory that all things must serve a purpose and therefore utilitarian (use) values of protected areas relate to the services and benefits that humans derive from biodiversity, such as products/ resources (direct use), knowledge, and ecosystem services which sustain our human life (indirect uses). These values are often what dominate in valuation studies. Non-utilitarian values (non-use) in valuation studies constitute existential values (intrinsic, cultural, heritage) and bequest values, which relate to the knowledge that others benefit or will benefit from a given ecosystem.

Marketable versus non-marketable values
Market valuation assigns a monetary value to an ecosystem or biodiversity or process that in theory could be compensated or traded. Some values can be captured by the financial market, for example products directly traded, whereas others, such as regulatory or cultural services, must be valued based on the costs of replacing such services (Alho, 2008). Non-marketable values relate to many of the same values considered intrinsic or non-utilitarian, such as cultural heritage or wellbeing, that are difficult to assign monetary value to.

1.3.3 Protected area values in the literature- trends

The literature review found that the number of studies on PA values has risen sharply in the last three years, after 2012. The five value groups were all well respresented in the literature with notably more emphasis on health and spiritual value and economic value. Of the 101 studies reviewed, 43 reported on more than one value. Those studies including a monetary valuation reported value generation in the areas of tourism, ecosystem services (e.g. carbon storage), or products (e.g. timber, food) (See Figure 1.3).

Figure 1.3: *Graphical summary of the trends in the literature on protected areas values: Top panel – the number of valuation studies conducted from 1977 to 2015 in the final dataset; Middle panel – the number of studies conducted on the respective value groups from 1977 to 2015; Bottom panel – the total number of studies conducted on the respective value groups. One study can be included in more than one value grouping, i.e. where multiple values are assessed or where a value can be considered relevant to more than one value grouping*

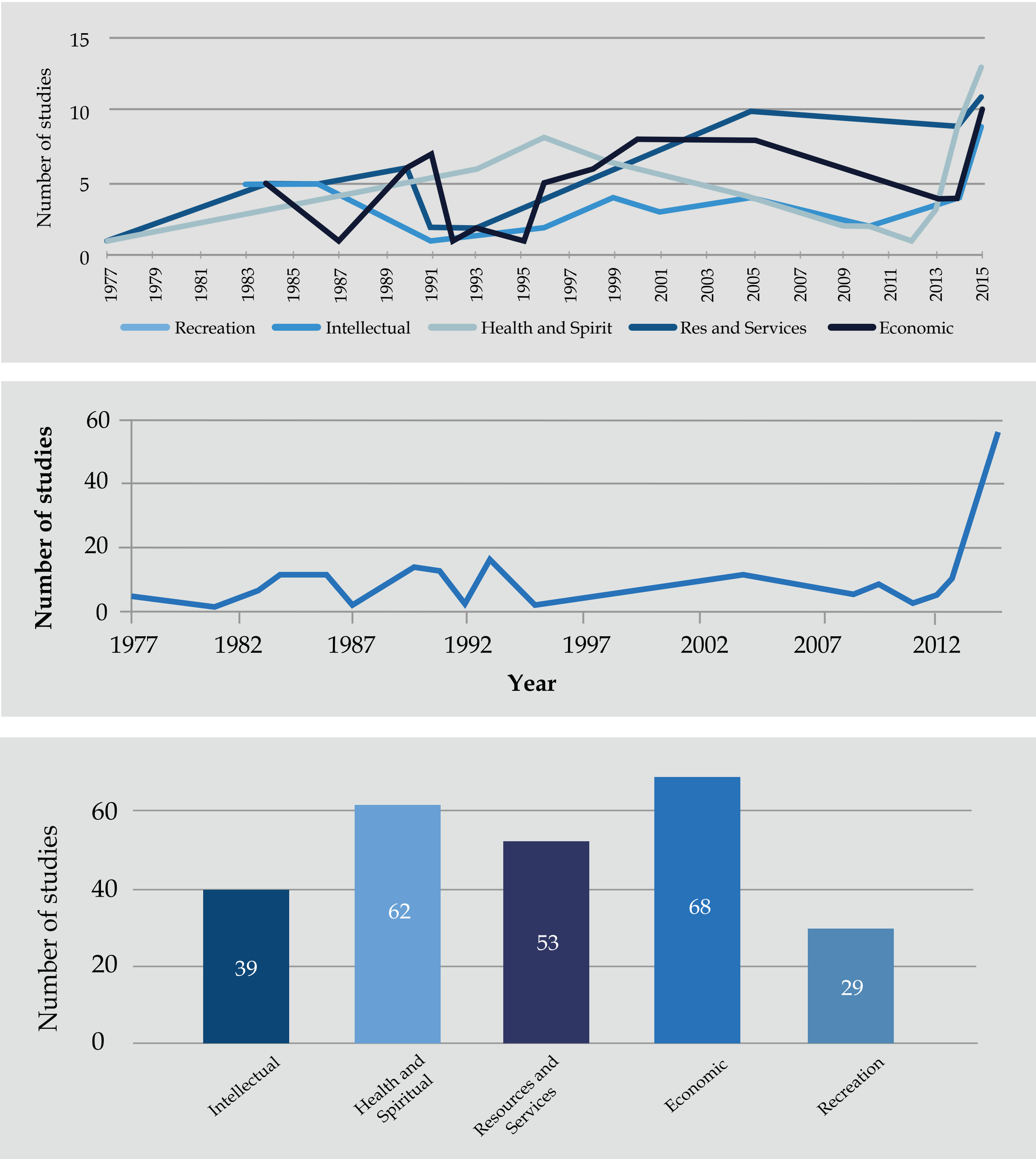
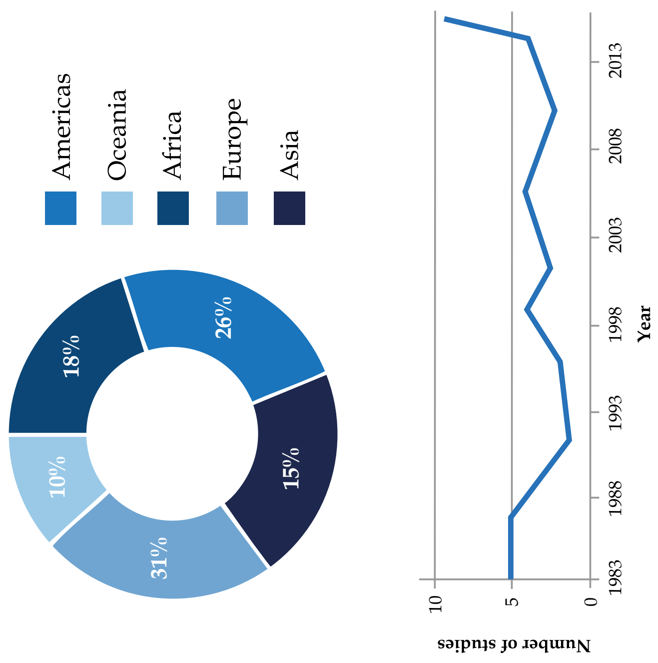
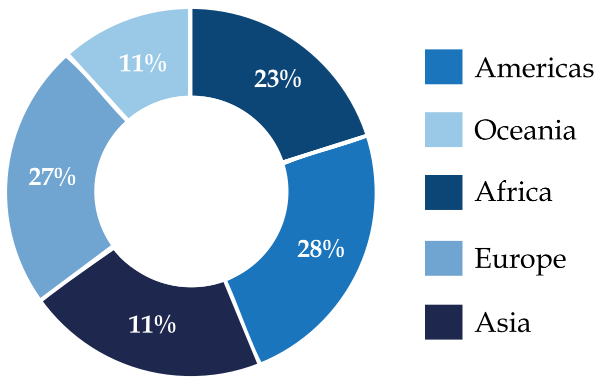
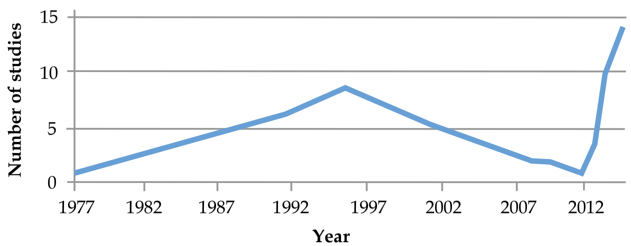
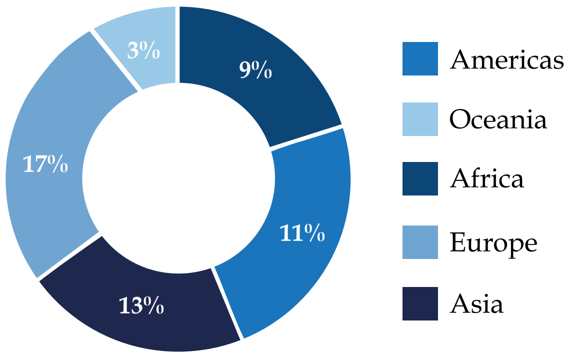
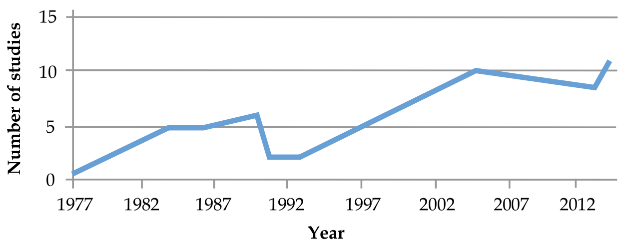
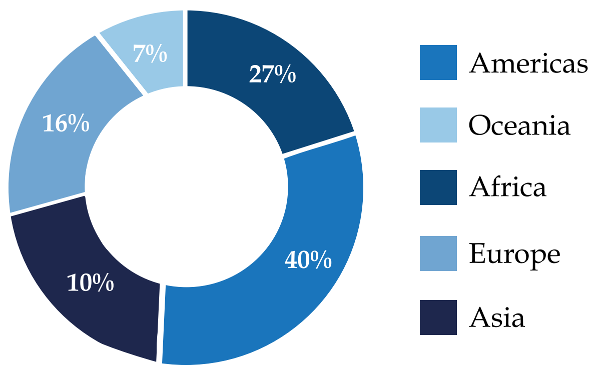
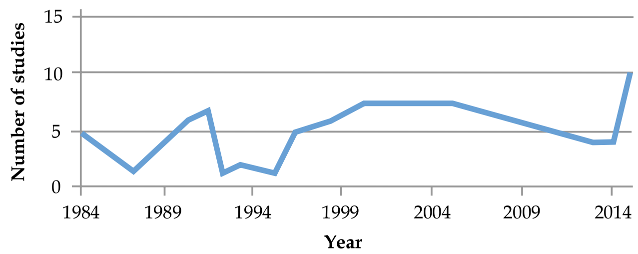


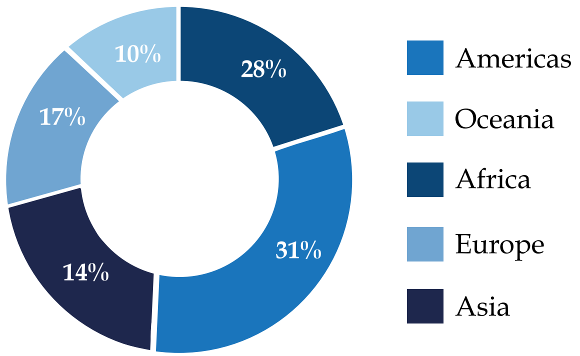
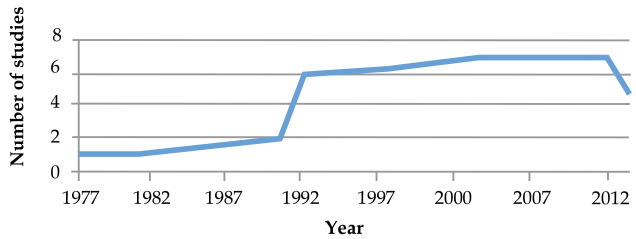
Table 1.1: *For each value group summary trends are shown: a pie chart shows the geographic trends, i.e. the percentage of valuations conducted within the grouping from each of the five regions (Africa, Americas, Asia, Europe, and Oceania). The line graph represents the number of studies included in the value group over time (using the 40-year timescale). Examples of the type of studies with the value focus of the group are given, and an average monetary value (US$/year) was calculated based on those studies for which a monetary value was provided.*

Value group	Trends	Examples	Average monetary value
Intellectual (N=39). This grouping includes studies which focus on scientific and learning values generated from PAs and/or nature.		Many of these studies attribute value to the biodiversity or conservation of a protected area. For example, the conservation value of Mediterranean wetlands (Spain) has been assessed using bird species assemblages as indicators of the impact of agricultural change for wildlife (Robledano et al, 2010). Similarly, the efficacy of China's national reserves is evaluated based on their benefit for migratory bird species (Yang et al, 2015).	Intellectual values are generally non-marketable and therefore most studies lack a monetary value. US$ 112,624 / year (39 valuations)

Value group	Trends	Examples	Average monetary value
Health and spiritual (N=62). This grouping includes studies which focus on intrinsic, spiritual, therapeutic, and cultural values generated from PAs and/or nature.	 	Such studies include those that focus on ecotourism benefits to visiting tourists but also socioeconomic and cultural values to local communities. A study from a reserve in Chile, with a focus on the conservation perception/behaviour relationship found that a positive perception of conservation by local communities, for example from increased social capital, does not necessarily translate into participation in conservation initiatives or prevent illegal resource use in reserves (Pickering et al, 2015). The non-market recreational benefits of Lundy Island Marine Reserve, UK were assessed using contingent valuation and travel costs of tourists visiting resulting in an estimated surplus of £359-574 (US$551-882) per trip (Chae et al, 2011).	These value types are more likely to be qualitatively assessed and less likely to be attributed a monetary value. US$57,043,530 / year (62 valuations)

Value group	Trends	Examples	Average monetary value
Resources and services (N=53). This grouping includes studies which focus on biodiversity and life sustaining values (e.g. ecosystem services, food) generated from PAs and/or nature.	 	In such studies the estimated value of an ecosystem service is often based on the cost for the equivalent service, or by using previous economic valuations from different regions, or by universal values such as carbon credits. For example, the valuation of carbon storage and sequestration of mangrove systems in southern Brazil resulted in an estimate of US$455,827 a year for sequestration and US$3,477,041 a year for storage; it considered pre-existing estimates in forest biomass plus the value of carbon credits (Estrada et al, 2015).	US$ 4.16 billion / year (53 valuations)

Value group	Trends	Examples	Average monetary value
Economic (N=68). This grouping includes studies which focus on direct and indirect economic return from PAs and/or nature.	 	In Africa PAs are justified as a valuable land use by estimating tourism spending and relating this to potential income from which local communities can benefit. In Kenya, Moran (1994) estimates a user surplus of US$407 million a year for the protected area network. Another study from Kenya considers the benefits of cooperative management of a reserve by local communities, NGOs and a private company whereby gross annual income of $65,000-140,000 (2002-2010) is reinvested in reserve and community (Lamers et al, 2014). Often a suite of values is assessed for a particular protected area, for example values are attributed for animal production, tourism, sewage treatment benefit and role in regional watershed within Ichkeul National Park, Tunisia totalling US$2,597,670 (Thomas et al, 1991).	US$ 270, 986, 922 / year (68 valuations)

Value group	Trends	Examples	Average monetary value
Recreation (N=29). This grouping includes studies which focus on recreation values generated from PAs and/or nature (e.g. ecotourism).	 	In valuing the mangrove areas in Sarawak, Malaysia, Bennett et al. (1993) estimated a revenue of US$7,061,069 a year from tourism as well as revenue generated directly from resource use (fisheries and timber), which total US$40,502,323. In Mozambique willingness-to-pay for marine-based tourism was assessed in the Ponta do Ouro Marine Reserve and an annual revenue range of US$108,311-244,755 was derived from the willingness-to-pay valuation.	US$ 5.01billion / year (29 valuations)

1.4 Valuation and Justification for Protected Areas

1.2.1 Development of the value of nature

As seen from the literature review above, nature valuation, including but not exclusive to protected areas, has developed over time; and a diverse range of values is considered and different methods used to estimate the value of protected areas and related services. At local and national scales (for which most studies are conducted), these assessments of value provide a baseline for monitoring the impact of development and other destructive activities on protected areas, and enable managers to assess the benefits of protected areas and identify shortcomings in resources in protecting assets. More generally, the studies in the literature add weight to the argument for maintaining well-managed protected areas, and requiring national and international policies to take account of their value.

Moreover, over the last 20 years a number of estimates of global ecosystem services have been made and these have highlighted the need to protect natural areas globally. Although these studies often rely on simplified values for components of complex natural systems, they provide a broader perspective on the importance of fully functioning systems and value. Summaries of key studies include:

Constanza et al (1997) estimate the value of 17 ecosystem services of 16 biomes using published studies plus original calculations, per unit area by biome then multiplied by total biome area and the sum of all services across all biomes. This resulted in an estimated global value average of US$33 trillion a year, compared to the annual global gross national product at the time of US$18 trillion. A recent paper by many of the same authors updated the estimate of the global value based on changes in ecosystem service value and land use. This increased the average annual value to US$125 trillion when converted to the same units (2007 US$), which equals an increase of $79 trillion on the previous estimate (Constanza et al, 2014).

Balmford et al (2002) similarly argue the need for evaluation of the numerous benefits humans derive from nature and protected areas to enhance the incentive to conserve nature. However they warn of the difficulty in using classic market-based economic valuation, as Constanza et al do (1997), which simply extrapolates value to a global scale, as this underestimates nature's value by not accounting for increased per unit demand as supply diminishes. Balmford et al (2002) therefore calculate the net marginal benefits (difference in benefit provision between intact and converted biomes). Overall they find the mean loss of total economic value due to conversion to be approximately 50%. Based on this they propose the overall benefit to cost ratio of effective global conservation to be at least 100:1

The increased emphasis on valuing nature is often a response to a divide between national policy for development and economic growth, and for protecting nature. Therefore to incentivise governments and policymakers, conservationists have promoted biodiversity and protecting ecosystems through the assessment of their monetary value and provision of services and resources, rather than their intrinsic value (Kareiva et al, 2012), as well as consideration of the loss of value through human impact. Indeed, the inclusion of nature as a value and tradable asset can now be seen in policy nationally and globally. For example, in Europe, the EU Directive on Environmental Impact Assessment introduced in 1985 made developers accountable for negative impacts on ecosystems and has since encouraged new tools and valuation methods across Europe to monetise the benefits of protected areas (Jones-Walters and Mulder, 2009).

More recently, the EU Biodiversity Strategy (2020) aims to halt the loss of biodiversity and ecosystem services and protect natural capital by incorporating protection into all other EU policies (e.g. agriculture, development, industry, tourism). It emphasises that the loss of biodiversity, 'has devastating economic costs for society which until now have not been integrated sufficiently into economic and other policies' (point 16) and incorporates the Economics of Ecosystems and Biodiversity (TEEB) project analysis and various other economic assessments from across the EU into valuations.

At the global scale, the Task Force on Economic Benefits of Protected Areas (1998) of the IUCN World Commission on Protected Areas (WCPA) calculates economic values of protected areas, and acknowledges the tension between leaving protected areas untouched and exploiting or developing them (differing degrees of multi-use). It attempts to follow the moves in policy to attribute value and benefit of protected areas to local and national communities and economies, and focuses on monetary, tangible values. More recently, TEEB was initiated in 2007 following a meeting of the environment ministers of G8+5 countries in Potsdam, Germany. It was proposed that the global economic importance of biodiversity, including costs of loss and benefits of protection be assessed. At REDD+ (reducing emissions from deforestation and forest degradation, and fostering conservation, sustainable management of forests, and enhancement of forest carbon stocks) is another initiative by the United Nations to assign financial value to the natural capital of nature, specifically forests, many of which are within protected landscapes. The purpose is to provide an incentive for reduction of deforestation and associated emissions.

At national scales, various initiatives have begun that also place nature in a market framework, whereby natural capital is tradable and given specific value. In the US, Wetland Mitigation Banking (EPA, 1995) is a form of environmental market trading where wetlands are developed to create marketable wetland credits (acres and function). These credits are sold to others as compensation for unavoidable wetland impacts. Similarly, in Australia, the Biodiversity Banking and Offsets Scheme (DECC, 2007) operates a market-based, voluntary programme that aims to counteract development activities that are detrimental to biodiversity by ensuring that they do not incur a net loss. It is an offset mechanism offering biodiversity credits to landowners. This focus on the banking and trade of nature removes previous restrictions and conservation of land potentially devaluing protected areas as a consequence.

1.4.2 Caveats on valuation

Different approaches to valuation yield large differences in outcomes and make it difficult for governments and policymakers to understand the true economic benefit of investing in PAs. The WCPA Task Force (1998) recommends the adoption of a standardised method to avoid protected area assets being undervalued. The valuation of nature is dependent on the method used and hypothetical market prices must often be calculated, making the value of ecosystems, biodiversity, and other services open to interpretation. There is the risk that conservationists and scientists will assign value with no sound basis as emphasis is increasingly given to the economic value of biodiversity, undermining standardised valuations and public and government trust of such methods (Lamb, 2013). This leads to possible underestimation, and if used as the basis of policy could be detrimental to ecosystems rather than beneficial.

The overarching issue is that nature and biodiversity are hard to standardise and value placed on them is subjective, whereas measuring monetary output is simpler and more objective. Justifying nature conservation and protected area value and resilience based on a merging of these two strands is challenging. What is clear, though, is that the values generated from PAs are large, diverse, and significant. Knowing who benefits from values generated by PAs, whether those values are monetisable and easy to quantify or non-monetisable and challenging to quantify, can help policymakers and PA managers to articulate the case that PAs should be placed ahead of other priorities. These arguments can be made directly to beneficiaries, whether communities, businesses, or society as a whole.

Section 2: Investments and creating value

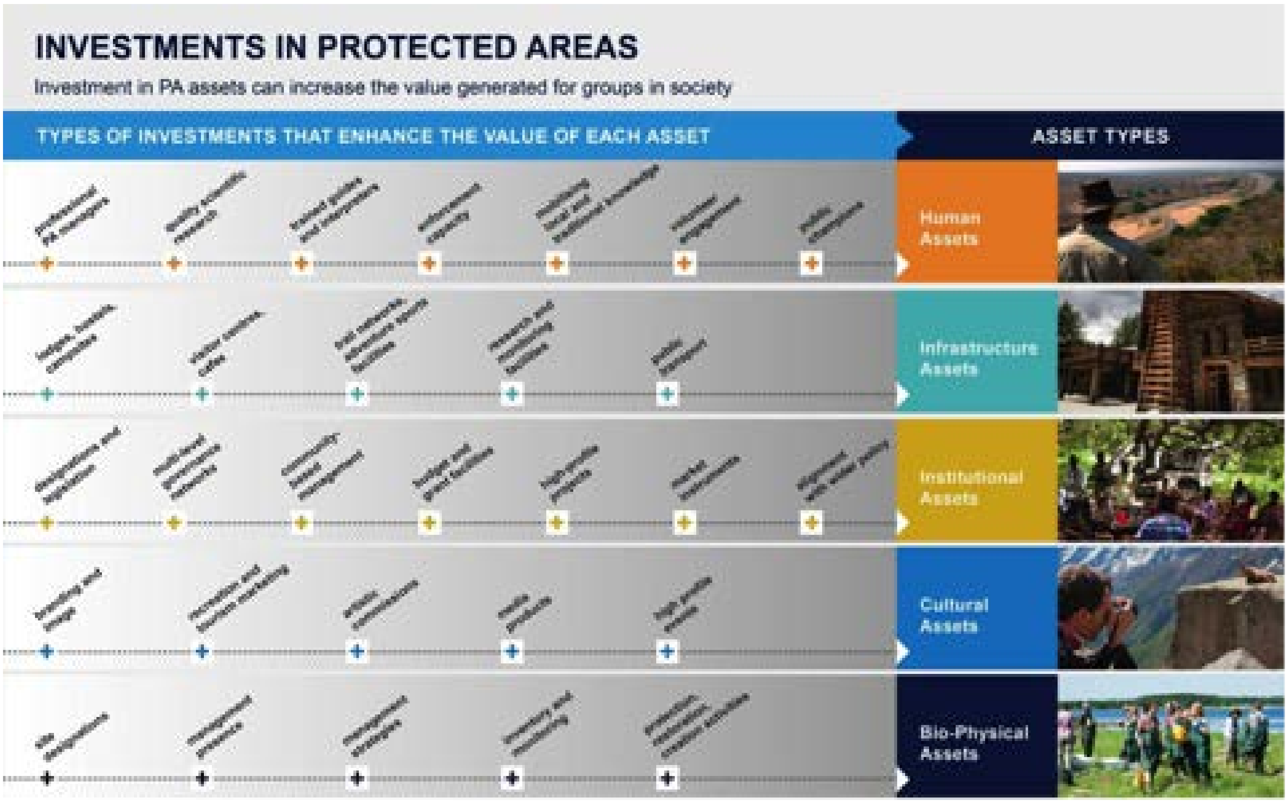
Authors: Ben Caldecott, Dexiang Chen, and Harriet Milligan

In the previous section, we systematically reviewed the literature on monetisable and non-monetisable values that PA assets generate and examined how views of value in a PA context have changed over time. This section examines the types of investment in PA assets that can generate value, what value, and for whom. We use four brief case studies to show how investment, value generation, and value capture are tied together: urban PAs of Manchester Mersey Valley, Ecological Station of Jataí, sustainable enterprise in Apuseni National Park of Romania, and outdoor recreation in Yellowstone National Park.

We also attempt to bring together data sources that help us to better understand how investment flows have changed over time, what the noticeable shifts in investment flows have been, and what may have caused these shifts. We then try to determine the state of PA funding and the size of potential investment gaps.

2.1 Investments in PAs, value generation and capture

Figure 2.1: *Investments in protected areas*



Traditional sources of operational and capital funding, particularly from governments and conservation organisations, have been insufficient to assure PA assets and are unlikely to increase at the scale or pace required. There is hence a need to demonstrate the various forms of value that accrue to different domains of society to reinvigorate the case for investment to a broad range of actors. Figure 2.1 above details the asset framework we have created to identify and specify the types of investment that enhance the value of the five asset types: human, infrastructure, institutional, cultural and biophysical.

Through four case studies set in different regions facing varying challenges, we illustrate the utility of such an asset framework in analysing the benefits that citizens, economies, enterprises and organisations gain through PA investments. We begin with the transformation of the Manchester Mersey Valley from a polluted industrial town into an urban protected area of enhanced human and institutional assets. Second, we demonstrate how the conversion of farmland into the ecological station in Jataí benefited citizens through the improvement of both human and cultural assets. The case study on the sustainable use of arnica (Arnica montana) harvesting in Apuseni National Park of Romania further demonstrates the potential application of market-based instruments. Finally, we look at the oldest national park, Yellowstone National Park, to show how investments into PAs focused on outdoor recreation generate value and the various stakeholders who capture it.

2.1.1 Manchester Mersey Valley – restored urban protected areas

Figure 2.2: *Map of Manchester Mersey Valley and the various infrastructures and developments*

Increased population of cities around the Mersey Valley in the 1900s led to the growth of engineering, chemical, and electrical industries. The rapid development of these industries throughout the 20th century resulted in huge environmental impacts on the Mersey River and its banks. Copious amounts of industrial effluent were pumped into the river; areas of poor agricultural land along the banks were set aside for rubbish tips and sewage works, and uncontrolled gravel extraction from the river for the building of motorways commenced. Stretches of the river were so polluted by sewage and industrial effluent that parts its fish life completely disappeared. The Mersey was thought to be one of the most polluted estuaries in Europe.

In the 1970s, Greater Manchester Council decided to invest some £1 billion in the Mersey Valley river corridor to convert the polluted river and its banks into an attractive 'countryside'. The river banks were raised and flood storage basins built, reducing the problem of flooding. Reclamation schemes were introduced, turning rubbish tips into agricultural land, converting sewage works into nature reserves and filling up gravel pits to form water parks.

Because of these investments, various forms of value were generated and captured by citizens, economies, enterprises, and organisations. These values and the entities that benefited are summarised in Table 2.1.

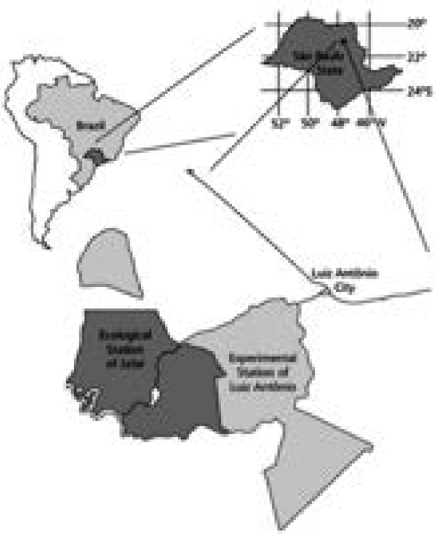
Table 2.1: *List of i) investment type ii) values generated, iii) the specific forms of values and iv) the entities that captured the values, as a result of Greater Manchester Council's investment in the Mersey Valley river corridor*

Investement type	Who captured value	Values Generated	Value Type
Sale water park	Citizens, economies and enterprises, organisations	Regional centre of excellence for water sports and angling	Status and prestige
	Citizens, economies and enterprises.	Increase housing prices	Economic flows
	Citizens	Creation of work placements and jobs	Quality livelihoods
Riverside walking trails	Citizens, organisations	Gatherings to observe nature	Outdoor recreations
	Citizens	Waymarked routes and guided opportunities for healthy wallking and accessing the natural attraction	Health and spiritual improvement
	Citizens, organisations,	Vibrant volunteer community supporting the Valley and its activites	Social cohesion and community building
Public transport	Citizens, economies and enterprises, organisations	Visitor attraction with great access via public transport links and a range of facilities which enhance the visitor experience	Economic flows, outdoor recreation
High profile projects and alignment with wider policy	Citizens, economies and enterprises, organisations, nation and polities	Green post-industrial city image	Status and prestige
	Citizens	Reduction of ecological impacts	Ecosystem services

2.1.2 Ecological Station of Jataí – research stations

The Ecological Station of Jataí – Conde Joaquim Augusto do Vale – located in the municipal district of Luiz Antônio of Brazil, has an area of 4,532.18 ha. Until the end of the 1950s, the area was heavily logged to extract wood for fences, construction of the railroad, civil construction (houses, school etc.), and canoes and furniture (Santos et al, 2000). In addition, locations near Sapé Lake were utilised for cultivation of rice, corn, potato and as pastures for domestic animals. Hunting for either sport or food by inhabitants of the municipal district area was also common (Santos et al, 2000). These activities resulted in unprecedented negative impacts, such as stream degradation, soil erosion, riparian forest degradation, introduction of invasive species, habitat fragmentation, and loss of biodiversity, threatening the rich biodiversity present in Jataí (Santos et al, 2000).

Figure 2.3: *Map of Ecological Station of Jataí*



In 1959, the Forest Institute of the state of São Paulo acquired the area, but it was only gazetted as an ecological station with the concern of preserving the remaining Cerrado vegetation and riparian forest in 1982. Grants from the Special Environmental Agency (SEMA, in the Ministry of the Interior) were utilised to set up infrastructure assets such as research and monitoring facilities and trail networks, and institutional assets such as community-based management programmes. This allowed for the subsequent development of programmes for universities and research institutions to pursue comparative ecological studies, in addition to the protection of biophysical values, which are under threat in Jataí (Nogueira-Neto and de Melo Carvalho, 1999). The investments also made possible the development of infrastructures such as living quarters for scientists, as well as a few other houses for guards and labourers (Nogueira-Neto and de Melo Carvalho, 1999).

Table 2.2: *List of i) Investment type ii) values generated, iii) the specific forms of values and iv) the entities that captured the values, as a result of the Forest Institute of the state of São Paulo's investment in Jataí Ecological Station*

Investement type	Who captured value	Values Generated	Value Type
Mobilising local and traditional knowledge	Citizens	Reduction of ecological impacts	Ecosystem services
	Citizens, organisations	Gathering of community for cultural activities and remembering historical events	Moral, aesthetic and scientific expression
High-profile projects and community-based management	Citizens, organisations, nation and polities	International recognition for curtailing deforestation	Status and prestige
Trail networks and facilities	Citizens, organisations	Gatherings to observe nature	Outdoor recreation
	Citizens, organisations	Opportunities for reflection, spiritual enrichment and cognitive development	Moral, aesthetic and scientific expression
Research and monitoring facilities	Citizens, organisations	Provide and inspirational and educative form of re-creative experience	Moral, aesthetic and scientific expression
	Citizens, economies and enterprise	Created jobs for people	Economic flows
Quality scientific research	Nation and polities, organisations, citizens	Improved scientific understanding in the biological and ecological aspects of tropical rainforests	Natural resources
	Nation and polities, citizens	Further knowledge on tropical rainforest aid in the establishment of national policies and technologies for using, controlling, and managing, the environment	Ecosystem services, Natural resources

2.1.3 *Apuseni National Park of Romania – sustainability of Arnica (Arnica montana) harvesting*

Apuseni National Park in Romania was created in 2004 to conserve its unique karstic nature. The park is populated by locals living in villages at high altitudes, with permanent and quasi-permanent dwellings (Bösze et al, 2014). The locals live a traditional way of life that depends heavily on the natural resources in the park (Bösze et al, 2014). However, in a study conducted by Albert Ludwigs University from 2000-04, the critically endangered medicinal plant, Arnica montana, was found to be over-harvested, at a low acquisition price and traded unfairly (Bösze et al, 2014).

WWF-UK, the Darwin Initiative, the WWF Danube-Carpathian Programme Office, UASVM Cluj-Napoca and Gîrda de Sus City Hall invested over US$300,000 in a project to identify a sustainable use of Arnica montana in the park for the benefit of both biodiversity conservation and the welfare of local people in the Apuseni mountains.

Figure 2.4: *Map of Apuseni National Park of Romania*

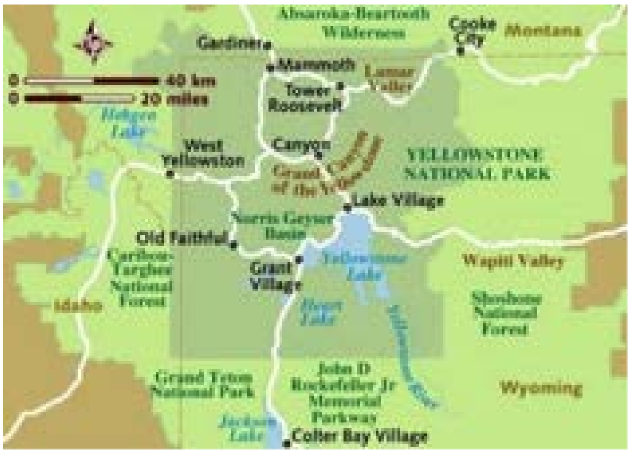
Because of the investment by the organisations, the Ecoherba Society, a local enterprise that dries organically produced arnica inflorescences, was established. In addition, the Ecoherba Society and German bio-cosmetics company, Weleda struck a contract deal.

Table 2.3: *List of i) Investment type ii) values generated, iii) the specific forms of values and iv) the entities that captured the values, as a result of WWF-UK, the Darwin Initiative, the WWF Danube-Carpathian Programme Office, UASVM Cluj-Napoca and Gîrda de Sus City Hall's investment in Apuseni National Park*

Investement type	Who captured value	Values Generated	Value Type
Establishment of local enterprise for sustainable use of Arnica	Citizens	Increased income and security of income for villagers	Economic flows
	Citizens, organisations	Ensure a sustainable population of Arnica montana	Natural Resources
High-profile economic projects and community-based management	Citizens, nation and polities, economy and enterprise	Product brand and supply-chain security	Status and prestige, Economic flows
	Citizens	Ability of citizens to treat their children's bruises with a natural remedy	Health and well-being
	Citizens	Presence of resident professionals in their community	Economic flows
Mobilising local and traditional knowledge	Citizens	Reduction of ecological impacts	Ecosystem services
	Citizens	Gathering of community for cultural activities and remembering historical events	Moral, aesthetic and scientific expression
Quality scientific research	Citizens, organisations	Improved scientific knowledge for the management of biodiversity in Apuseni	Moral, aesthetic and scientific expression
	Citizens	Training and development of local young professionals	Capacity building
	Nation and polities, organisations, citizens	Presentations and sharing at international conferences	Status and prestige

2.1.4 Yellowstone National Park– Outdoor recreation

Figure 2.5: *Map of Yellowstone National Park*



Established in 1872, Yellowstone National Park is arguably the very first area to be legally protected from settlement, occupancy or sale (Merrill, 2003). It is widely believed that the park was established as the result of an 1870 expedition that revealed the unique physical and biological aspects of the area, motivating the protection and setting apart of Yellowstone National Park as a public space or recreation ground for the benefit and enjoyment of Americans into the future.

Since then, the US government has invested an estimated US$30-55 million annually in lodging facilities, outdoor activities, sports and recreation, environmental education programmes, developing cultural and historical attractions, and maintaining the scenic vistas and biodiversity.

The National Park Service adopts a business plan management model that charts investments and their returns annually, allowing for clear and detailed monitoring of current park operations, funding and park priorities and funding strategies (MPS, 2015). Figures from 2014 indicated that an estimated cumulative benefit of US$543 million was obtained from the approximately US$50 million congressionally appropriated investments spent on Yellowstone National Park (NPS, 2015). These investments allowed for the maintenance of trails and roads for visitors, facility operations, education programmes as well as the management of cultural and natural resources. In turn, various forms of values were generated and captured. For instance, 67,000 jobs were created as a result, encouraging economic output in local gateway economies surrounding Yellowstone National Park (NPS, 2015). An expanded list of the forms of values generated by these investments in outdoor recreation is detailed in Table 2.4.

***Table 2.4:** List of i) Investment type ii) values generated, iii) the specific forms of values and iv) the entities that captured the values, as a result of the US government's investment in Yellowstone National Park*

Investement type	Who captured value	Values Generated	Value Type
Professional PA managers	Citizens	Job provision and security of income for locals	Economic flows
	Nation and polities	International recognition for management excellence	Status and prestige
Trail networks and facilities	Citizens, organisations	Gatherings to observe nature	Outdoor recreation
	Citizens, organisations	Provide an inspirational and educative form of re-creative experience	Moral, aesthetic and scientific expression
Lodges, campsites, hotels	Citizens	Job provision and security of income for locals	Economic flows
	Citizens, organisations	Gatherings to observe nature	Outdoor recreation
Branding and image	Nation and polities	National and regional identity	Status and prestige
	Nations and polities, Economies and enterprise	Foreign exchange via tourism	Economic flows
Designation and legislation	Nation and polities	National and regional identity	Status and prestige
	Nations and polities, Economies and enterprise	Foreign exchange via tourism	Economic flows
	Citizens, nation and polities, economy and enterprise	International recognition as a place of special historic, cultural and natural significance	Status and prestige, Economic flows
Quality scientific research	Nation and polities, organisations, citizens	Improved scientific understanding in the biological and ecological aspects	Natural resources
	Nation and polities, citizens	Further knowledge on tropical rainforest aid in the establishment of national policies and technologies for using, controlling, and managing, the environment	Ecosystem services, Natural resources
	Nation and polities, organisations, citizens	Presentations and sharing at international conferences	Status and prestige

2.2 Investment trends in protected areas

2.2.1 Global data on investment flows into protected areas

To understand where investments to generate more value should be made and prioritised, it is necessary to understand the relationships between protected area investments, the forms of value they generate, and who or what captures this value. Earlier in this section and in previous sections we have outlined elements of a framework that could achieve this.

We now attempt to use historical data to apply our framework to real PA resource decisions. This may enable us to better understand i) where previous investments in protected areas were made and whether this reflects certain preferences or factors prevalent in the conservation community; ii) how investments have performed over time and how much value was generated; and iii) what forms of value investments generate over time – do investments create expected values or not, and are these sustained? Understanding these issues could have significant implications for future investment decisions within PAs, PA networks, and across the global PA estate.

Unfortunately, even for global financial flows for biodiversity conservation, studies and information are sparse (Parker et al, 2012; Waldron et al, 2013). Information on financial flows to protected areas is even more limited, with bulk of the available literature in the form of policy guides on funding sources (IUCN, 2000), manuals to set up sustainable financial schemes (Bovarnick et al, 2010), or instructions on developing financial plans (Flores et al, 2008). With a diverse array of funders having taken an interest in protected areas over many decades, there is a need for an up-to-date source of information on the funding landscape in protected areas.

2.2.2 Aim of study

To help close this knowledge gap, we conducted an extensive search for data over several months. This was not exhaustive, but allowed us to marshal sufficient data sources to conduct a high-level initial assessment of historical, location-specific, and investor-specific investment trends in the global protected area estate. Anticipating the likely difficulties in obtaining information, we also sought to identify the key data challenges and provide suggestions for future in-depth analysis.

Specifically, we aim to answer the following questions:

- How much money has been invested in protected areas globally?
- Who are largest funders of protected areas?
- How have investment trends changed over time, for example, between PA categories, asset types, and regions?
- Which regions are the largest recipients of those funds?
- Are there changes in regional focus over time?

2.2.3 Methods

2.2.3.1 Data acquisition and limitations

Our study looked at regional investment flows of protected areas by conducting an extensive search of secondary and 'grey' literature such as financial reports, NGO reviews, as well as scientific journals. We extracted and compiled all available information on financial inputs and their sources, dating back as far as possible.

2.2.3.2 Data selection

PA projects are often embedded within the wider scope of biodiversity projects. We screened all projects carefully and only selected those that are directed specifically at PAs, or carried out, conducted around or within them.

2.2.3.3 Resultant data

212 records detailing investment flows in 43 countries from six regions were identified and used for the global analysis of investment flows into PAs. Long-term financial data appears to be severely limited accounting for less than 5% of the records.

2.2.3.4 Assumptions

The analysis is based on the following assumptions:
Category overlap: as we are gathering investment inputs from various sources, there are instances where investments in PAs and programmes are repeated by co-investors. We assume that the investors are reporting figures from their organisation only and calculated the total regional investment by adding up all individual amounts.

Missing reports: there are some projects for which the final approved sum is missing. In addition, a handful of reports with figures from previous years had values that did not tally. In such cases, we accorded priority to the value presented in the latest report (for instance, if values reported for 2012 differed in the 2013 and 2014 records, we assume the 2014 record is correct).

Geography: funds are categorised as regional even when the funds are shared by just two countries. As a result, investments in certain countries may be underestimated.

Timing: for this study, investments were compared using the year they were approved on the assumption that funds were disbursed in the same year. However, actual timing for project implementation and disbursement of funds may vary, since projects often span several years and may encounter delays. In some cases, the approved amounts are never fully disbursed or projects fully implemented. Due to this variability, anticipated funding and project intent at the time of approval is the most consistent indicator across funders, even though it may result in a slight overestimation of actual expenditures.

2.2.3.5 Challenges and Limitations

A global survey relying on secondary literature inevitably has limitations and omissions. However, the inherent weaknesses and gaps in data clearly demonstrate the acute need to collect PA investment data on a regular and standardised basis. Comparisons of conservation investments between countries, the identification of priority areas for funding, and the assessment of global financial flows will continue to be elusive without such efforts.

The study is limited in depth and breadth with less than half of all the countries contributing to global protected area estate included. Some of the excluded countries, due to absence of data in secondary sources, include important areas of species richness and endemism such as China and Russia. Another area of weakness is the level of detail in the information provided by individual sources. Furthermore, information is weak particularly in the regional distribution of investments in PAs.

At present, there is no mechanism for collecting standardised annual data on investments to protected areas. This study found assessments inconsistently spread over a number of different years, and many developing countries were omitted from the samples. Attempts to collate long-term government funding data for PAs was challenged by the lack of publicly available data that isolated PA components from national budget statistics. PA budgets in different countries are often reported together with other conservation and environmental operations in a single budget. Despite our best efforts to clarify the figures, this may have led to cases of both underestimates and overestimates. As a result of the severe constraints to the data, few, if any, firm conclusions can be drawn.

Nevertheless, the study serves well as a review of existing literature, provides a broad, updated overview to global investment flows into PAs, and elucidates recent trends in the various sources of PA funding. Comments, criticisms and alternative calculations of investment flows from readers will be highly appreciated to aid in the improvement of our understanding. Future studies should consider obtaining primary data from PAs. This could be achieved by the development of a national reporting protocol for PA investments and a regularly updated, accessible database, which would facilitate access to more reliable and current data for an improved overview of historical, location-specific, and investor-specific trends.

2.2.4 Global overview of investment flows into protected areas

At a global level, our research revealed a paucity of information on PA finance – only eight studies. Recent studies have all cited figures from the only comprehensive global survey of PA finance conducted 20 years ago and published by UNEP-WCMC (James et al, 1999). The most recent mention of global investment flows into PAs by Balmford et al (2015) recognised the imprecision of utilising dated data by providing an equivocal estimate of <US$10 billion.

The majority of the studies focused on the investment necessary to maintain and manage existing PA systems. These studies have estimated the amount of global investment flowing into PAs as ranging from US$1.1 billion to <US$10 billion (Table 2.5). These studies have however failed to consider the funds utilised to expand PA networks. The only two studies that have considered investments devoted to the creation of additional PAs estimated the annual amount invested in PAs at $US13 billion (Table 2.5).

Considering the upper limit of global investment flows into PAs, the amount of funds going into PAs constitute more than half of the estimated annual global investment in biodiversity conservation projects (Figure 2.6). However, it represents less than 1% of the total annual funds set aside for sustainable development projects. This suggests biodiversity conservation and PAs are low on the list of funding priorities compared to other sustainable development initiatives.

Six main categories of funding sources contributed to investment flows into protected areas (Figure 2.7). There is evidently a considerable reliance on public investments for protected area projects. Domestic government budgets were the main investors, followed by multilateral funds. Together, they accounted for over 90% of the annual funds that went into protected areas. Development aid agencies were the next, contributing an annual average of US$0.4 billion. The amount injected into projects by NGOs, private and philanthropic foundations were the lowest, accounting for only 3% at an average annual amount in total.

***Table 2.5:** List of studies on global investment flows into PAs*

Focus of study	Amount	Amount inclusive of	Reference
Survey of 600 PA agencies on the management costs and costs of creating of new PAs	US$6B/yr	Management only	James et al., 1999
Modelled data based on protected-area size, with adjustments for national income and purchasing power	US$2.3/yr	Management only	Balmford et al., 2003
Based on expenditure on existing reserves and the opportunity costs of existing and additional reserves	US$13B/yr	Management and expansion	Balford and Whitten, 2003
Based on protected-area size and fixed ratios to other management components	US$1.1B/yr (Developing countries)	Management only	Vreugdenhil et al., 2003
Combination of previous models and new data	US$13B/yr	Management and expansion	Bruner et al., 2004
Citing James et al., 1999	US$6.5B/yr	Management only	Emerton et al., 2006
Citing James et al., 1999	US$6.5B/yr	Management only	Lopoukhine et al., 2012
Citing James et al., 1999	<US10B/yr	Management only	Balmford et al., 2015

Figure 2.6: Proportion of global investment in sustainable development (blue), biodiversity projects (grey) and protected area projects (green)

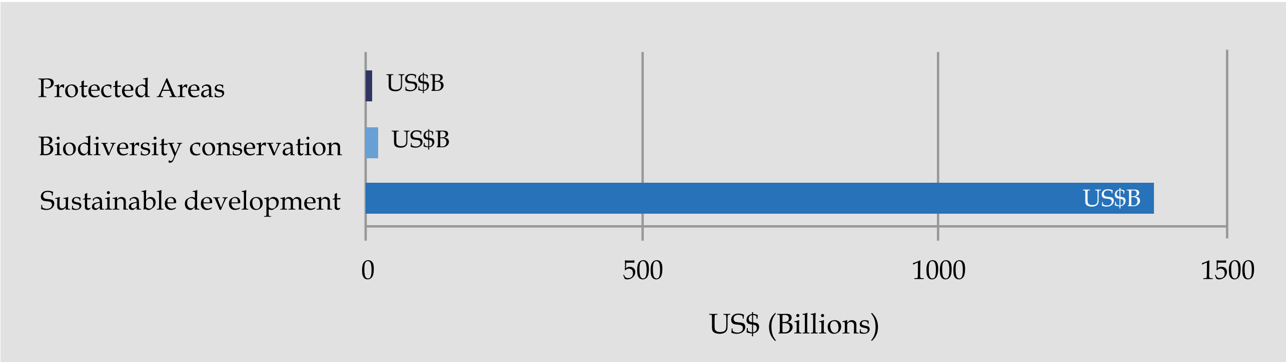
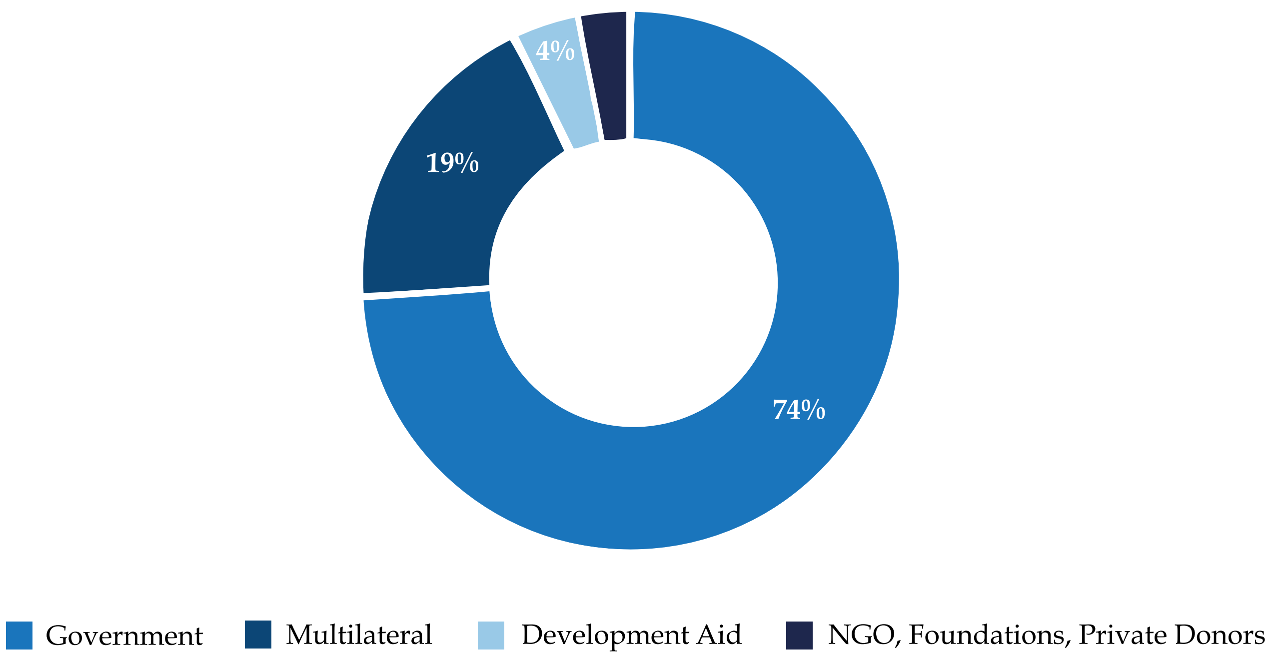


Figure 2.7: Sources of investments and the proportion of their contribution to protected area projects



Footnotes:

Government data obtained from Mansourian and Dudley (2008), GEF and development aid data obtained from Emerton et al. 2006), NGO, foundations and private donor amount estimated using percentage of overall biodiversity conservation reported in Waldron et al. (2012).

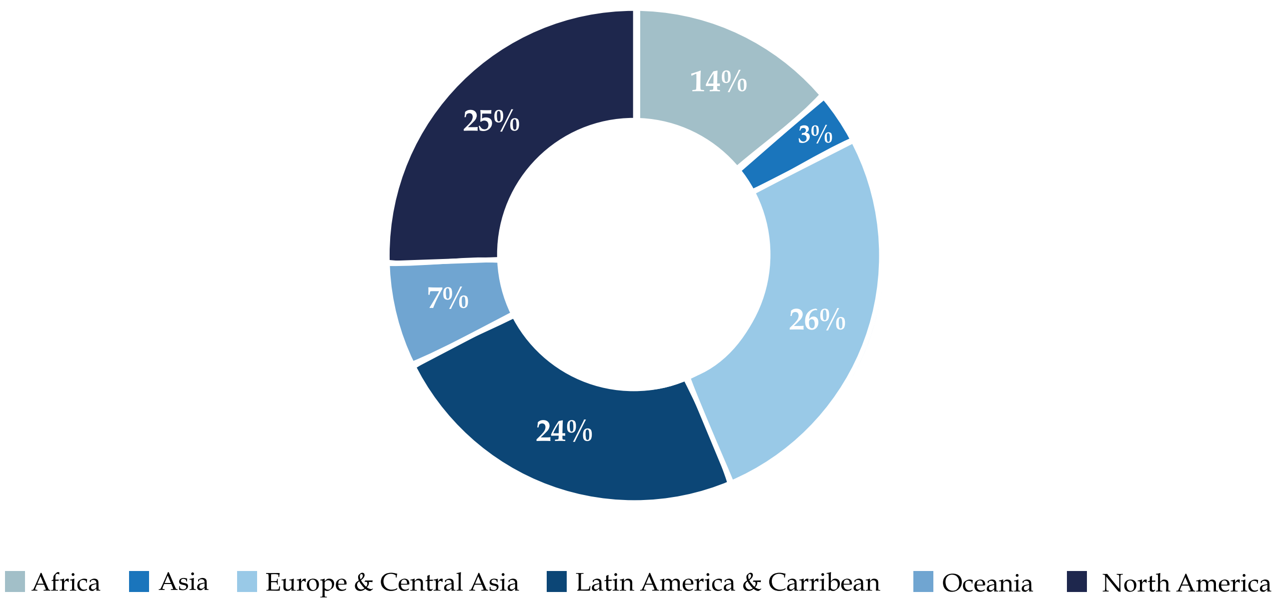
2.2.5 Regional outlook on investment flows into Protected Areas

The amount of investments flowing into developed and developing countries appears to be equal, with North American PAs and PAs in the Europe and Central Asia regions receiving US$7 billion a year in total, accounting for over half of the annual global investment funds (Figure 2.8). Countries in these regions, such as the United States and the United Kingdom, generally have higher GDPs and have historically invested significantly in PAs. In addition, member states of the European Union are also committed to the Natura 2000 network and hence to invest in the creation and management of PAs in their countries.

PAs in the Latin American and Caribbean regions received investments amounting to US$3 billion a year channelled towards protected area management and development every year, while Africa receives a considerably smaller share of the global investment of US$1.8 billion per year (Figure 2.8). Since the 1980s, the need to address poverty while promoting sustainable development in biodiversity-rich areas has been much discussed. This was formalised in the Integrated Conservation and Development Projects (ICDPs) presented at the 1992 Rio summit (Franks and Blomley, 2004). Realising that without addressing the critical factor of poverty initiatives to promote either conservation or development were doomed to failure, ICDPs proposed a holistic view taking into consideration both conservation and development needs simultaneously (Alpert 1996, Sanderson and Redford 2003, Sanderson 2005). These ICDPs facilitated the transfer of substantial funds from developed to less developed countries particularly in Africa, Latin America and the Caribbean (Romero et al, 2012).

Asian and Oceanian regions receive the lowest amount of global investment for PAs – less than US$1 billion per annum respectively (Figure 2.8). This is possibly due to the insular nature of many states, which makes it harder to both manage and coordinate protected area investments. In addition, the concentration of emerging economies in these regions means an abundance of investment opportunities with shorter and higher returns on investment.

Figure 2.8: *Proportion invested in respective regions*



2.2.6 Sources of PA finance

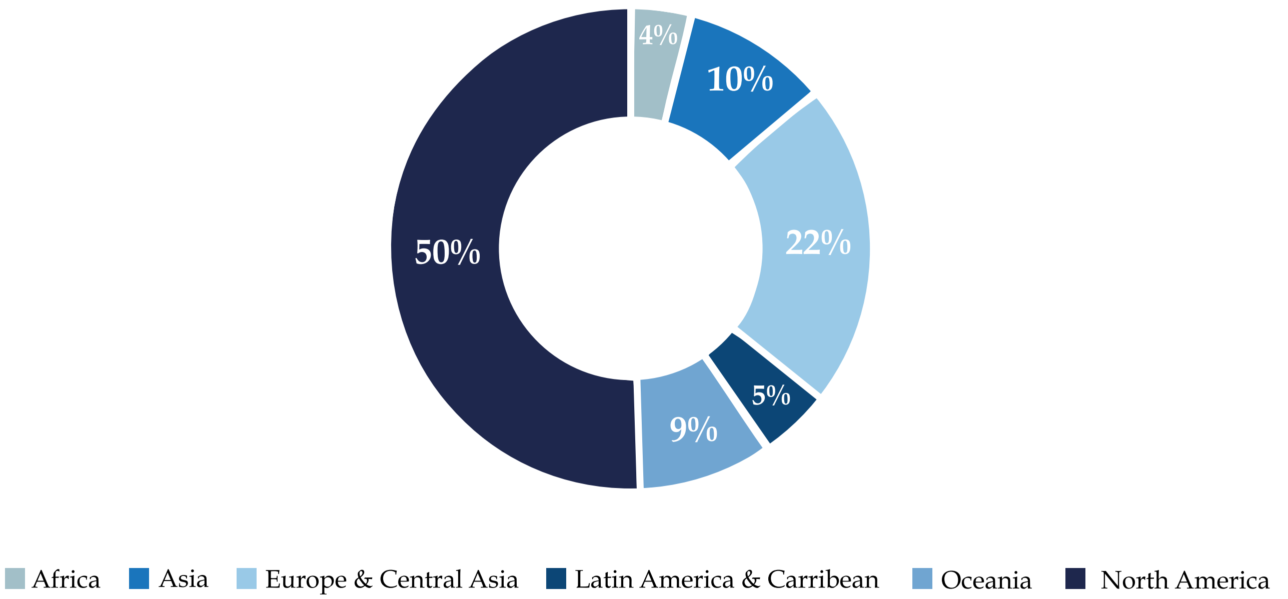
2.2.6.1 Domestic government budgets

Domestic government budgets are the largest funding source for PAs globally. Likewise, they are also typically the largest source of PA financing in most countries (Emerton et al, 2006). Our study utilised an updated dataset from the study conducted by Mansourain and Dudley (2008) across six regions to investigate the geographical distribution of domestic government budgets.

The amount of domestic government spending on PAs is highly variable, ranging from US$0.0028/ha/yr in Bolivia to US$1665/ha/yr in Bolivia. High government spending is clearly skewed towards the wealthier developed countries, with North American PAs and European PAs receiving considerably more government funds than poorer countries (Figure 2.9). This is consistent with previous studies, which revealed that in developed countries government funding covers a larger proportion of PA budgets.

Long-term data obtained on the US federal government budget for the National Parks Service provides an insight into trends in government funding over time (Figure 2.10). Government spending on PAs has increased relatively steadily from the mid-1980s over 15 years but has been notably volatile in the past decade. There has been a gradual decline following the 2008 financial crisis, which is consistent with media reports (Beamish, 2013). The data supports Watson et al's (2014) study, further demonstrating that governments have potentially been sliding back on commitments to PAs in recent years.

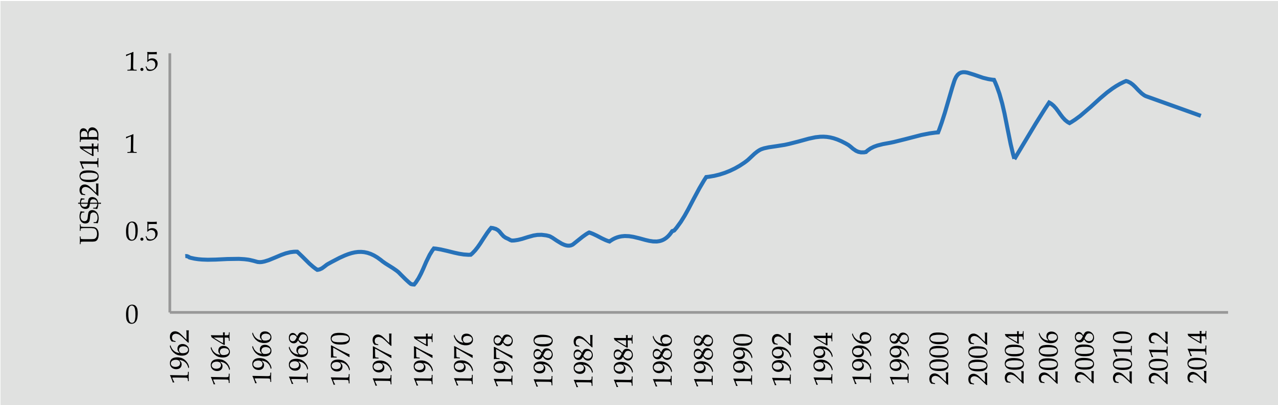
Figure 2.9: *Distribution of government investment flowing into PAs in respective regions*



Footnotes:

Data sources for North America: (Dudley et al, 2007); Asia: (Dudley et al, 2007); Africa: (James et al, 2003; Dudley et al, 2007); Oceania: (Dudley et al, 2007); Europe and Central Asia: (Gantioler et al, 2010); Latin America: (McElhinny, 2007)

Figure 2.10: *US Federal Government budget for the National Parks Service (2014 US$billions) from 1962 to 2014*



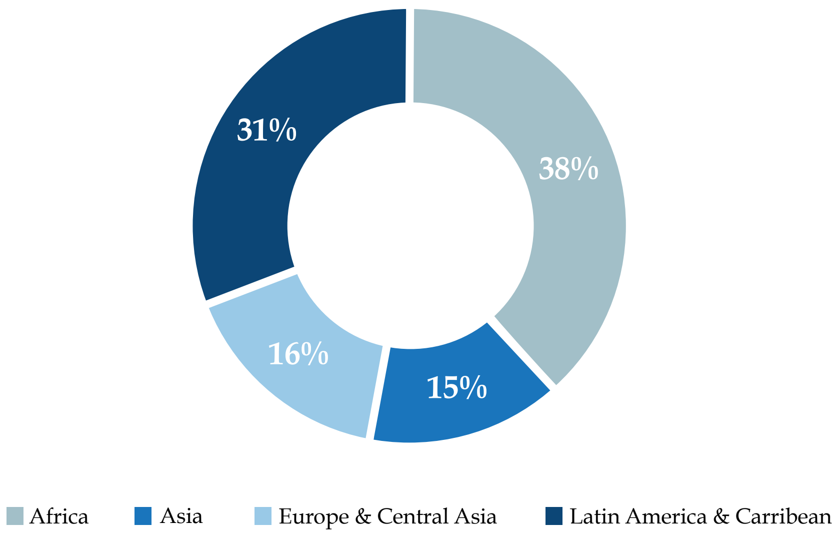
Source: www.usgovernmentspending.com/federal_spending_chart

2.2.6.2 Multilateral funds

Contributing an annual investment of more than US$2.5 billion towards PAs globally, multilateral agencies are their second largest investment source. They are especially critical for PAs in the developing world, which typically have limited domestic government investments (Figure 2.11). The main direct source of multilateral finance into PAs comes from the Global Environment Facility (GEF). Together with the co-financing it leverages, GEF provides another estimated US$1.1 billion towards PAs annually.

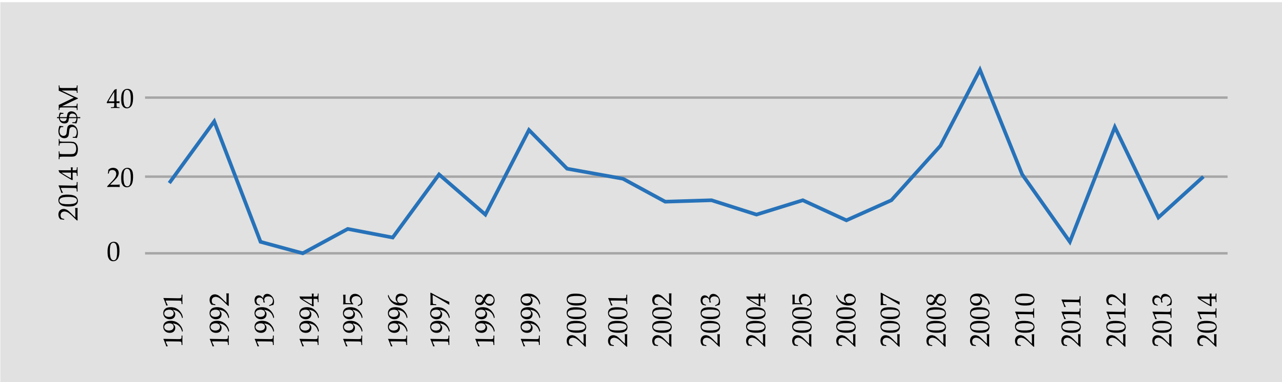
The geographical distribution of multilateral funds are notably biased towards Latin America, the Caribbean and African countries, accounting for almost 70% of annual multilateral investments in PAs (Figure 2.11). This could be due to the high number of Integrated Conservation and Development Projects (ICDPs) in Africa, Latin America and the Caribbean, which are providing additional funding towards PAs (Romero et al, 2012).

Figure 2.11: *Distribution of multilateral investment flowing into PAs in respective regions*



Annual GEF funding data from 1991 to 2014 revealed that the amount is highly variable, fluctuating between US$1 million to US$45 million (Figure 2.12). This large variation in annual funding is typical of multilateral funds, which follow a system to identify priorities, allocate funding, release tenders, grant opportunities and award contracts.

Figure 2.12: *GEF investments in protected areas (2014 US$ millions) from 1991 to 2014*

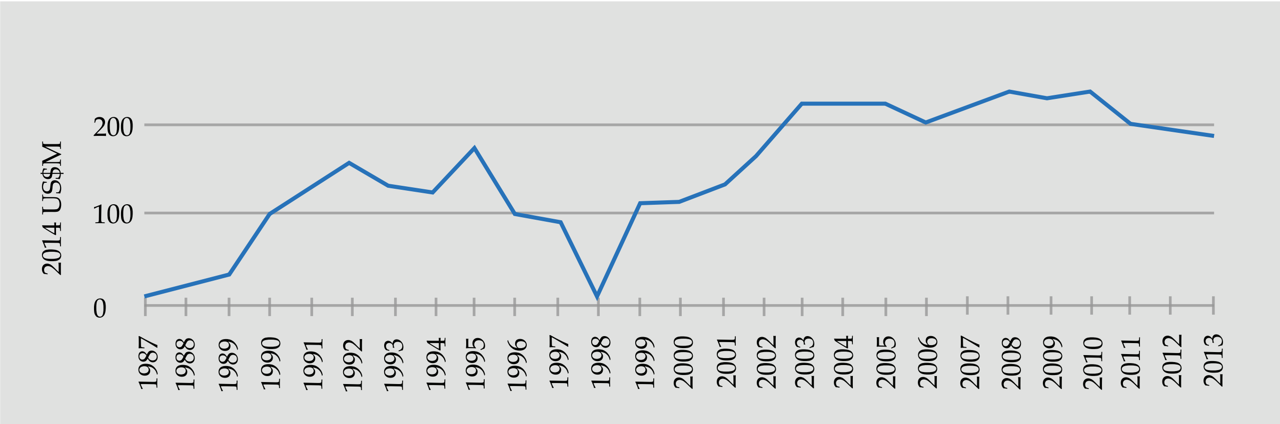


2.2.6.3 Development aid

Although development aid represents a fraction of global investment in PAs, it is arguably the next most important source of funding for PAs in developing countries after multilateral funds, disbursing an estimated US$300 million a year for biodiversity projects. The International Development Association (IDA) is part of the World Bank aimed at reducing poverty in the poorest countries by providing credits and grants. IDA projects provide an additional US$250 million a year to the poorest countries to help them maintain their existing PAs (Emerton et al, 2006). Regional development banks, both the Asian Development Bank and the Inter-American Development Bank invest a further US$250 million a year and US$500 million a year respectively in PAs as part of the sustainable development and poverty alleviation projects (Emerton et al, 2006). Furthermore, 3% of development assistance provided by the European Union is directed towards biodiversity conservation and sustainable use, amounting to just under US$200 million annually. Although only a fraction of these funds is channelled towards PAs, collectively they amount to half of the annual multilateral funds flowing into PAs around the world.

The amount of funding targeted at PAs in different regions is not clear from the data available. However, time series data from a major development aid provider is shown below in Figure 2.13. USAID is one of the few donors where such time series data was readily available. The annual amount invested in PAs ranged from US$10 million to US$200 million. There was a decline in USAID investments in PAs from 1995 to 1998 that corresponds with the return of a Republican Congress, which sought cuts in spending, particularly in family planning and environmental protection (Hicks et al, 2008), and increased emphasis on development assistance, food aid and economic support funds instead (German and Rande, 1998). There was also a decline from 2010 to 2013, which corresponds to the reduction in biodiversity-related aid over the same period reported by Watson et al (2013).

Figure 2.13: *USAID investments in PA projects from 1987 to 2013 (2014 US$ millions)*



Source: https://www.usaid.gov/results-and-data/budget-spending

2.2.6.4 NGOs, philanthropic funding, and investments from businesses

In addition to playing an important role in the governance of conservation and PAs, environmental NGOs have also provided substantial financial support to PAs. However, detailed global analysis was not possible due to a lack of consistent reporting. Brockington and Scholfield (2010) provided the only global estimate of NGO expenditure on PAs and this estimated an annual amount of US$200 million. Looking through individual NGO accounts and annual reports yielded only sporadic and incomplete information.

Funding for PAs also comes from private sources, including individual donors, philanthropic foundations, and local communities. For instance, the United Nations Foundation provides an average of US$10 million a year to fund World Heritage Sites. The Gordon and Betty Moore Foundation has also provided grants amounting to US$261 million over ten years in support of establishing protected areas.

Records detailing private sector expenditure on PAs was extremely limited. Nonetheless, businesses appear to be showing growing interest in PAs and this is evidenced by a growing number of donations in recent years (Business in the Community, 2011). In addition, businesses are gradually opening up and supporting the development of mechanisms that can support PAs, such as results-based payment mechanisms, ‘green’ bonds, and biodiversity offsets (Business in the Community, 2011). However, there is a lack of information on private sector investment in PAs resulting from these new structures and mechanisms, many of which are still in their infancy.

The absence of multi-year data for NGO, philanthropic funding, and private sector investments into PAs, and the challenges of aggregating a meaningful sample of datasets means that our understanding of these funding sources is extremely weak. This should be a priority for further data development, particularly through consistent and comparable disclosures from funders of PAs.

2.3 Identifying potential protected area investment needs and opportunities

2.3.1 Global trends of investment needs and opportunities

Although there has not been any study specifically of investment gaps and priorities in the protected area landscape, we believe that the global study of management effectiveness evaluation conducted between late 2005 and 2010 by Leverington et al (2010) can provide valuable insights on the subject.

The global study conducted by Leverington et al (2010) on Protected Area Management Effectiveness (PAME) examined the management effectiveness of protected areas of 140 countries and assessed if projects, interventions, and management activities were achieving their aims and how they could be improved.

A total of 5,878 management effectiveness assessments were collected in the study by Leverington et al (2010). As various organisations employed differing assessment criteria and matrices, the authors had to first standardise them into a common reporting format. The final format has 35 defining indicators representing major themes and elements. The respective management standard for each individual indicator was measured on a scale from 0 (no management standard) to 1 (highest management standard). These scores were compiled and mean scores across all studies were computed for each indicator.

Pearson's Correlation Coefficient analysis was performed to investigate if the 35 indicators used in PAME correlated with the management effectiveness of protected areas. To achieve this, Leverington et al (2010) tested the strength of the correlations between these individual indicators and the overall average score for management effectiveness and subsequently ranked them. This ranking provides a sense of which interventions might yield the most reliable or significant improvements in PA management.

There are a number of caveats to this analysis. First, the conversion of assessments into a common format may have led to the loss of some information. The qualitative nature of the assessments means that the original scores are highly dependent on the knowledge of evaluators and might contain personal biases. Finally, there are observable sampling biases. For example, the bulk of PAME assessments were undertaken by non-governmental conservation organisations and tend to be biased towards PAs that are perceived to be particularly vulnerable (Leverington et al, 2010). Also, more extensive and comprehensive records were reported in developed countries in North America and Europe as compared to developing countries in Africa and Asia (Leverington et al, 2010). Nonetheless, considering the substantial sample size, the study performs well as an exploratory analysis of general trends.

On the basis of our ranking, we have identified three key components of protected areas from the PAME correlation analysis that should be a priority for investment.

1) Resource Management and Monitoring

Both the PAME study and investment flows research revealed a shortfall in reliable, long-term global and regional data. The identification of the PAs and PA asset types in need of investment requires more comprehensive, up-to-date spatial and temporal information at global, regional and local scales. This can be achieved via the establishment of common reporting standards and frameworks and new databases.

2) Facilities, equipment, and infrastructure

It is evident from the correlation analysis that effective PAs are strongly linked to the adequacy of infrastructure, equipment, and facilities. Investments in infrastructure are needed in new PAs to meet new demands and to help ensure permanence (Dudley, 2008). Established PAs on the other hand require facilities and services to accommodate use by visitors and staff. This is especially critical in PAs promoting ecotourism which requires additional infrastructure for information, transport, accommodation, food, safety and recreation, in order to meet visitor expectations and generate higher user fees (Eagles, 2004).

3) Protected Area Mangement Planning

The PAME correlation analysis revealed three key areas of PA management planning which are strongly related to desired outcomes 1) design and establishment, 2) engagement and community involvement, and 3) proactive management capacity. Investments to support PA managers develop more robust management plans, especially in early stages of establishment, are important. Funding should also be disbursed to more established PAs that are underperforming.

2.3.2 Regional trends of investment needs and opportunities

All countries were shown to have a fair share of both well-managed PAs as well as poorly managed areas. However, there appear to be regional differences in management effectiveness. PAs with the least effective management are located in the Asian and Oceanian regions (Figure 2.14), which also receive the least in global funds for PAs (see Figure 2.8).

Correlation analysis of PAME indicators with effectiveness scores showed regional variation in investment need and opportunities. Table 2.6 illustrates the top indicators that correlated to PA effectiveness in the six identified regions. PAs located in regions which scored poorly in the PAME assessment (Asia and Oceania) appear to be in need of stable, long-term investment particularly to create, improve and maintain human, infrastructure and institutional assets.

PAs situated in regions with moderate PA effectiveness (Africa, Latin America and the Caribbean) appear to rely heavily on community benefit and assistance schemes and visitor management. Hence, targeted investments into human and cultural assets will create the specific values necessary to improve PA effectiveness in these regions.

Table 2.6: *Overall correlation of indicators with protected area effectiveness. Rank 1 suggests an indicator with high correlation while Rank 35 suggests no discernible correlation to protected area management effectiveness. Note: Ranking does not suggest importance.*

Indicator	Overall Rank
Adequacy of infrastructure, equipment and facilities	1
Effectiveness of administration including financial management	2
Communication program	3
Adequacy of relevant, available information for management	4
Adequacy of staff training	5
Management planning	6
Adequacy of hr policies and procedures	7
Research and monitoring of natural/ cultural management	8
Achievement of set work program	9
Adequacy of building and maintenance systems	10
Visitors catered for and impacts managed appropriately	11
Security/ reliability of funding	12
Management effectiveness evaluation undertaken	13
Adequacy of current funding	14
Appropriate program of community benefit/ assistance	15
Adequacy of law enforcement capacity	16
Effectiveness of governance and leadership	17
Staff morale	18
Adequacy of staff numbers	19
Constraint or support by external political and civil environment	20
Involvement of communities and stakeholders	21

(Continued on next page)

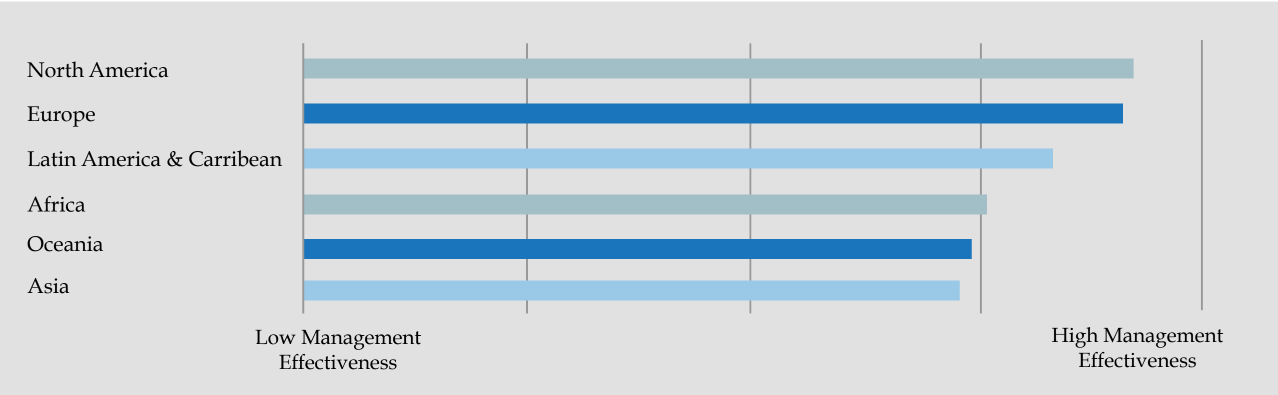
Table 2.6: Continued from previous page

Indicator	Overall Rank
Adequacy of PA legislation	22
Results and outputs have been produced	23
Threat monitoring	24
Sustainable resource management	25
Appropriateness of design	26
Marking and security/ fencing of park boundaries	27
Proportion of stated objectives achieved	28
Staff/ other management partners skill level	29
Conservation of nominated values -condition	30
Natural resource and cultural protection activities	31
Conservation of nominated values - trend	32
Effect of park management on local community	33
Park gazettal	34
Tenure security and issues	35

Footnotes:
Overall ranks based on values reported in Leverington et al (2010)

Finally, the investment needs in regions with the highest effectiveness scores (Europe and North America) appear to be less discernible. More specific analysis of investment needs at country or state level is required considering the lack of a clear trend presented at a regional level.

***Figure 2.14:** Relative PA management effectiveness across different regions*



Scores calculated based on regional correlation values reported in Leverington et al (2010)

***Table 2.7:** Top five indicators that coincided with PA effectiveness. Rank 1 suggests an indicator with high correlation while Rank 5 suggests no discernible correlation to protected area management effectiveness. Note: Ranking does not suggest importance*

Region	Rank	Indicator
Asia	1	Visitors catered for and impacts managed appropriately
	2	Adequacy of current funding
	3	Adequacy of staff numbers
	4	Proportion of stated objectives achieved
	5	Adequacy of PA legislation
Oceania	1	Adequacy of current funding
	2	Adequacy of staff training
	3	Adequacy of infrastructure, equipment and facilities
	4	Security/ reliability of funding
	5	Adequacy of hr policies and procedures
Africa	1	Security/ reliability of funding
	2	Appropriate program of community benefit/ assistance
	3	Adequacy of current funding

Table 2.7: *Continued from previous page*

Region	Rank	Indicator
	4	Visitors catered for and impacts managed appropriately
	5	Adequacy of building and maintenance systems
Latin America & Caribbean	1	Management effectiveness evaluation undertaken
	2	Adequacy of staff numbers
	3	Security/ reliability of funding
	4	Adequacy of current funding
	5	Appropriate program of community benefit/ assistance
Europe	1	Appropriate program of community benefit/ assistance
	2	Adequacy of current funding
	3	Adequacy of staff numbers
	4	Visitors catered for and impacts managed appropriately
	5	Adequacy of infrastructure, equipment and facilities
North America	1	Adequacy of current funding
	2	Effect of park management on local community
	3	Conservation of nominated values -trend
	4	Research and monitoring of natural/ cultural management
	5	Appropriate program of community benefit/ assistance

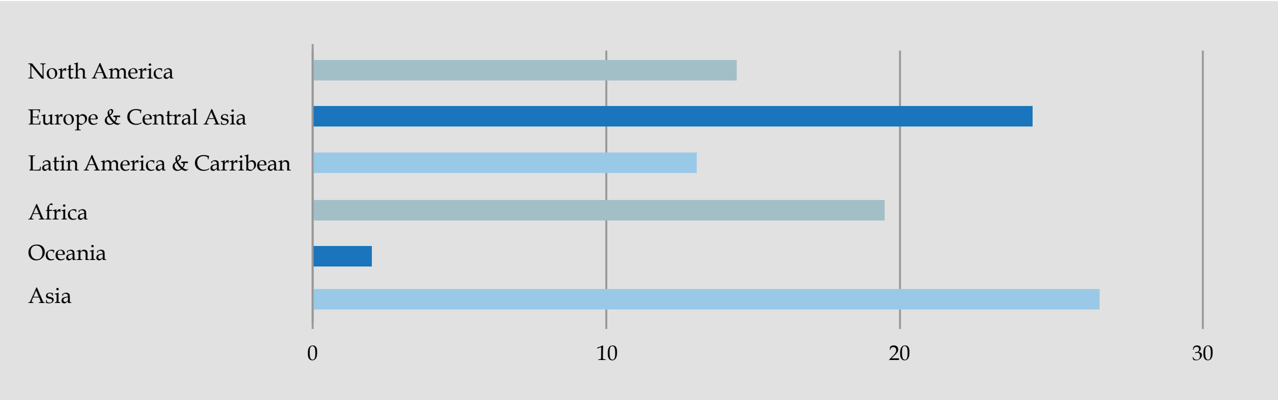
Ranks based on values reported in Leverington et al (2010)

2.3.3 Shifts in PA investments since 1999

Although establishing regional investment needs and opportunities is important, it is equally crucial to understand how investment trends in PAs respond to identified funding gaps. Utilising the seminal study on global PA investments and shortfalls conducted by James et al (1999) and comparing it with data obtained in the current study allows us crudely to assess if funds have been channelled into areas with identified funding needs.

In 1999 the highest funding gaps were identified in Asia, Europe and Central Asia, and Africa, while Oceania had the smallest funding shortfall (Figure 2.15). There appear to have been considerable shifts in investment flows since 1999, but they do not appear to be strategically directed to PAs based on their shortfalls. The proportion of funds for Asia, for instance, has decreased despite being clearly recognised in 1999 as the region which required the most financial help, while Latin America and the Caribbean have seen significant increase in investments in their PAs despite their comparatively low financial shortfalls (Figure 2.16). Meanwhile, although PAs in North America were assessed to have moderately high financial shortfalls, global investment to them has almost halved since 1999.

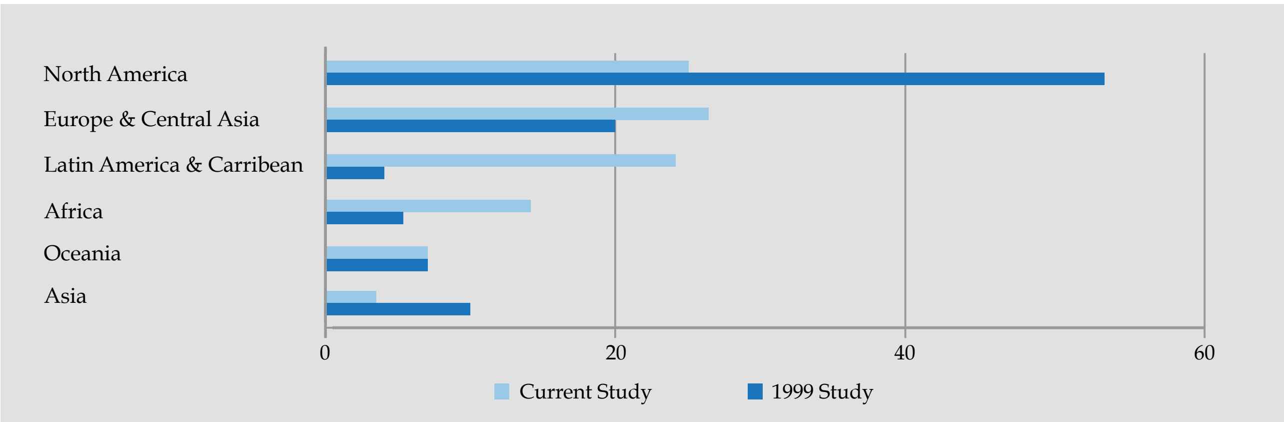
***Figure 2.15:** Relative PA funding shortfalls across different regions reported in 1999*



Analysis of investments in global biodiversity by Waldron et al (2013) revealed that factors such as threatened biodiversity, land area, costs, GDP and political stability explained funding trends and patterns better than reported financial shortfalls. Considering the funding sources PAs share with biodiversity conservation, their investment flows are also likely to be affected by these factors.

It is clear, however, that current available data is too sparse to conclude much with certainty. Moreover, given that multiple factors determine shifts in investment flows, there is a need for micro-level analysis of trends in specific PAs. These detailed analyses can be found in Part I, where we take a closer look at case studies of PAs in Tanzania and Brazil.

***Figure 2.16:** Shifts in proportion of global investment in PAs across different regions*



Section 3: Threats, materiality, and risk management

Authors: Ben Caldecott, Dexiang Chen, Harriet Milligan, and Natalie Page

3.1 Assessment of threats and their materiality

In the previous section, we reviewed the types of investment in PA assets that can generate value, what value, and for whom. We also attempted to bring together data sources to help us to better understand how investment flows have changed over time, what the noticeable shifts in investment flows have been, and what may have caused these shifts. We also attempted to determine the state of PA funding and the size of potential investment gaps.

This section reviews the diverse threats facing PAs and PA assets, the materiality of these risk factors (in terms of likelihood, timing, and size of potential impact), and the risk management options that might be available to policymakers and PA managers. PAs are coming under increasing pressure due to a number of threats, and these risk factors are impairing their ability to create value (see Figure 3.1 below).

Figure 3.1: *Value-at-Risk and risk management options*

3.1.1 Breaching Ecological Thresholds

3.1.1.1 Introduction

The concept of an ecological threshold refers to the change from one state to another, often the shift from one stable zone to another through passing a breaking point (Radford and Bennett, 2004). Over the last few decades, both ecologists and economists have investigated the existence of these thresholds in natural and modified ecosystems in order to develop conservation tools for the sustainable management of natural resources (Huggett, 2005). These alternative states may occur in response to anthropogenic disturbances, such as intense deforestation or increased nutrient loads resulting from sewage inflows and new urban development (Scheffer et al, 1993). The switch between alternative states can be sudden around the threshold value (Huggett, 2005, Wissel, 1984). A collection of research scientists from social and ecological disciplines known as 'the Resilience Alliance' defines an ecological threshold as a 'bifurcation point between alternate states which when passed causes a system to "flip" to a different state' (Huggett, 2005, Meyers and Walker, 2003). A characteristic of biodiversity loss is that it is associated with ecological threshold effects. (Perrings and Pearce, 1994)

3.1.1.2 Historical context and geographical distribution

Within the life sciences, the concept of thresholds has been investigated since the late 18th century (Huggett, 2005). In the context of ecosystems, the 'practical definition' (Huggett, 2005) of ecological thresholds was developed by Radford and Bennett (2004), suggesting ecological thresholds are the points or zones at which relatively rapid change occurs from one ecological condition to another.

Research on ecological thresholds compares a 'baseline' stable state – a formative natural state to refer to for conservation management purposes – to all possible novel or non-novel stable states. For example, the land area of the United Kingdom before human settlement was a forest-dominated system. Large-scale human alteration occurred over thousands of years, and many ecologists propose that certain types of pastoral landscapes can be considered as British 'baselines' (Gillson et al, 2011, Willis et al, 2010).

Novel ecosystems are hybrid systems retaining some original characteristics of an ecosystem, alongside novel elements, often anthropogenic influences, such as anthropogenic climate change, land use change, overexploitation or introduced species (Hobbs et al, 2009). The number of novel ecosystems around the world is growing, as direct and indirect human impacts are already widespread and still increasing in protected areas. Novel ecosystems now cover almost twice the global area of land than that covered by traditional forms of wilderness (Ellis, 2013). This creates new demands for effective conservation and the management of biodiversity found in novel ecosystems.

3.1.1.3 Materiality

Human activities have the capacity to push numerous fragile protected area ecosystems past critical thresholds, into qualitatively different stable states. Examples of protected areas which could face imminent threats of experiencing breaching thresholds or state shifts include the potential collapse of the Atlantic thermohaline circulation, Amazon rainforest dieback, and the loss of the Greenland or Arctic ice sheet (Lenton et al, 2008). The breaching of these thresholds will lead to a significantly novel ecosystem and a new stable state. Cumulative smaller changes can ultimately lead to long lasting consequences. A combination of high deforestation rates in the Amazonian rainforest with a projected warmer, drier climate from the perspectives of land hydrology could lead to a savannah-like ecosystem (Davies-Barnard et al, 2015, Pokhrel et al, 2014, Poulter et al, 2010, Malhi et al, 2009, Huntingford et al, 2008). Carbon cycle models of 'business-as-usual' emissions scenarios predict a rapid loss of Amazonian rainforest from the 21st century and onwards (Pokhrel et al, 2014).

Amazonian dieback will not only have a devastating impact upon the biodiversity that exists within the three-dimensional structure of the rainforest, but will also lead to risks of cultural losses, such as the loss of traditional ecological knowledge (TEK) from Amazonian indigenous territories (Walker et al, 2015).

In Australia, as a second example, hypotheses of habitat loss and fragmentation predict thresholds will be breached for the viability of native species populations, such as lizards, birds and marsupials, which will compromise the viability and longevity of these populations in the wild (Huggett, 2005; Sarre et al, 1995; Van der Ree et al, 2004).

Species diversity may significantly affect ecosystem resilience and, as a general rule, the more diverse an ecosystem, the more resilient it remains across time (Primack, 2014). There may also exist differing thresholds for individual species with, for example, different habitat requirements and niches or degrees of mobility (Huggett, 2005). For example, multiple species of forest birds do not fly through open spaces over a certain threshold. In the Western Australian wheatbelt, isolation thresholds for sedentary woodland birds was around 1,500m, as above this figure the birds did not disperse to remnant native vegetation in a highly fragmented landscape (Brooker et al, 1999). Other species are affected by conditions such as moisture and hydrological thresholds, habitat loss or modification, population isolation, or mutual links between other species within a complex ecosystem network (Huggett, 2005).

3.1.11.4 Manageability

As previously discussed, ecosystems typically respond nonlinearly to anthropogenic stressors and small changes in a stressor could result in large changes in ecosystem state and delivery of ecosystem services. Hence, it is imperative for PA management to have detailed understanding of the characteristics of the ecosystems within the PAs, such as baseline states, trophic cascades and system dynamics. This knowledge is essential in order to monitor and maintain systems proactively, preventing tipping points beyond which reversing ecological changes can be costly or, in many cases, even unfeasible.

Detailed review and analysis of 51 studies on ecological thresholds over the past three decades by Kelly et al (2014) also highlighted the importance of scale. Their research emphasised the critical importance of monitoring over temporal and spatial scales relevant to the ecological threshold (Kelly, 2014). PA managers, especially those working in PAs covering extensive areas and multiple ecosystems, should hence consider local and regional level management rather than decision-making at larger spatial scales.

Finally, successful management also requires policies that trigger action in response to scientific monitoring data. This will require PA managers to engage local communities, stakeholders and, local and national governments in policy discussions and to draft response actions, based on the possible scenarios, where ecological thresholds are breached.

3.1.2 Climate Change

3.1.2.1 Introduction

Climate change is a risk factor that threatens protected areas due to its large-scale impact upon the planet. The unpredictability of its regional impacts, of the responses of species, and of rainfall patterns accentuates the challenges faced by protected area managers. The inability to control or mitigate the effects of climate change results in an increased need for impact management strategies, based upon a combination of the predicted behaviour and responses of species, and of predicted weather patterns and the occurrence of natural disasters, fires and droughts. Even with strategic management, however, it is likely that climate change will cause a loss of species and result in damage to protected areas from natural disasters. However, effective management has the potential to mitigate this loss significantly.

Climate change has been recognised as one of the greatest threats to protected areas (e.g. Thomas et al, 2004; Hannah et al, 2008; Hole et al, 2009; Loarie et al, 2009). It has the capacity to impact protected areas in a number of ways, including through mass biodiversity loss, disease outbreaks, drought, flooding, fires, species range shifts, and loss of habitat (e.g. Parmesan et al, 2003; Thomas et al, 2004; Loarie et al, 2009). Table 3.1 (below) lists the possible impacts of climate change, how these affect protected areas, and what these impacts mean to human societies.

The impact upon different biomes and specific protected areas is likely to vary extensively due to geographical location, biotic composition and local climatic and topographical features (Stern, 2007; Loarie et al, 2009; Houghton, 2009). Terrestrial and marine protected areas (MPAs) will also suffer different effects. Protected areas and protected area networks will require tailored management strategies that take account of the range of climate change scenarios.

3.1.2.2 Geographical distribution and materiality

Predictions of the global average surface temperature provide relatively little information about regional implications. The way that atmospheric circulation operates, and the interactions that govern the behaviour of the whole climate system mean climate change across the globe will not be at all uniform (Solomon et al, 2007; Houghton, 2009). However, it is the local and regional changes that will be most felt by both human societies and natural ecosystems (Stern, 2007; Houghton, 2009). A complex network of changes may result from changes in temperature and precipitation, and ecosystems will respond very differently to these changes. Yet despite the more unpredictable nature of local impacts it is still possible to discern likely continental impacts and project how these may affect individual nations (Houghton, 2009). Figure 3.2 (below) depicts the predicted likely effects of climate change upon each major biome type, and where these biomes are distributed across the planet. Systems most threatened by climate change include coral reefs and atolls, mangroves, boreal and tropical forests, polar and alpine systems, prairie wetlands, and remnant native grasslands (Houghton, 2009).

Table 3.1: *Possible effects of climate change on protected areas*

Terrestrial Protected Areas	
Risk	Impact on Protected Areas
Impacts on forests	About 30% of the Earth's surface is natural forest, and 1-2% of this is plantation forest. Changes in climate can alter the suitability of a region for different species, and change their competitiveness within an ecosystem, so that even relatively small changes in climate will lead to changes in the composition of an ecosystem over time (Houghton, 2009). Trees and forests are particularly sensitive to the average climate in which they develop. In experimental and interactive climatic models of the Amazon rainforest, extremely high CO2 scenarios lead to a positive climate feedback. Instead of increased plant growth, the model experiences a dieback of Amazonian forest and a significant release of carbon into the atmosphere. As the forest dies back, rainfall is reduced because of changes in the land surface properties, leading to a new stable state of semi-arid (savannah-like) conditions rather than forest (Cox et al, 2004). Such results may also be associated with El Niño events, according to model simulations (Cochrane, 1999).
Fire risks	El Niño events, which are expected to become more frequent with climate change, can increase droughts and fires in South East Asia and the Amazon. Tropical forests which have experienced few or no forest fires previously will be at risk of increased fire frequency, as well as drought and loss of soil moisture. Tropical forests are maladapted to fires and the recovery rate is poor (Cochrane, 1999; Nepstad, 1999; Siegert et al, 2001; van der Werf et al, 2004).
Loss of fresh water resources	Increased human populations and a desire for higher living standards have increased the demand for freshwater. Many regions close to the sea are being affected by the withdrawal of groundwater, and the rise in sea level will add to this problem (Houghton, 2009).
Loss of land with sea level rise	Half of humanity inhabits coastal zones around the world. Within these, the low-lying areas are some of the most fertile and densely populated. Examples include Bangladesh, the Netherlands and small low-lying islands in the Pacific and other oceans (Houghton, 2009).
Mass biodiversity loss	Significant disruptions of ecosystems from disturbances such as fire, drought, pest infestation, invasion of species, storms and coral bleaching events are expected to increase. Such stresses on ecological systems combined with climate change threaten substantial damage or complete loss of unique ecosystems and biomes, and the extinction of endangered species. Extinction diminishes the diversity and complexity of life. With climate change and other human activities, the extinction rate is exceeding the speciation rate, resulting in an overall loss of genetic diversity, species diversity, and ecosystem complexity. Evidence exists to show that extinctions can alter key ecosystem processes and affect biological sustainability and productivity (Hooper et al, 2005; Wardle et al, 2011). Biodiversity loss is a significant driver of ecosystem change. If enough living connections are broken, whole ecosystems could collapse (Hooper et al, 2012).
Habitat range shift	Climate is the dominant factor determining the distribution of biomes (Scott, 2009). Natural ecosystems will become increasingly unmatched to their environment. With changes in climate over a few decades many ecosystems will not be able to respond as fast, or species migrate as fast. Fossil records indicate that the maximum rate most species have migrated in the past is 1km per annum. Without human intervention many species cannot keep up with the rate of movement of their climate niche, even without barriers to their movement imposed by land use. Plant and animal ranges are likely to shift poleward and upward in elevation for plants, insects, birds and fish (Houghton, 2009). Invasive species may also become more of a threat than they already are. Species in search of new niches are more likely with range shifts, encouraging competition. The increased global travel of humans exacerbates this problem.

***Table 3.1:** Continued from previous page*

Terrestrial Protected Areas	
Risk	Impact on Protected Areas
Disease outbreaks	Climate change has the capacity to increase the prevalence of a number of threatening diseases damaging to rare species, including chytridiomycosis, a chytrid fungus (Batrachochytrium dendrobatidis) that is a major killer of many amphibian species in neo-tropical communities (Lips et al, 2005). Higher global temperatures encourage mosquito-borne diseases, their transmission rates and geographical ranges, although human activities and global movements are a significant factor in this spread (Reiter, 2001). Plant pathogens have high adaptive potential, particularly with climate change, and epidemics are a concern for numerous plant populations (Garrett et al, 2006).
Carbon dioxide (CO2) 'fertilisation' effect	One effect of increased atmospheric CO2 is the boost to growth in plants. Higher CO2 concentrations stimulate photosynthesis, enabling plants to fix carbon at a higher rate. Although this means numerous plants can grow significantly faster under experimental conditions (e.g. Reilly et al, 1996), under real-life conditions where water and nutrients are limiting factors, growth quality declines with CO2 enrichment and higher temperatures. Susceptibility to pests and diseases may also be affected (Houghton, 2009).
Increased rainfall and flooding	Precipitation is expected to increase in northern high latitudes in winter and the monsoon regions of southeast Asia in summer (Houghton, 2009).
Drought and desertification	Southern Europe, Central America and southern Africa are expecting significantly drier summers. An increase in temperature will mean a higher proportion of the water falling in the Earth's surface will evaporate, even in regions with increased precipitation. In regions with unchanged or less precipitation there will be substantially less water available at the surface. Combined effects of less rainfall and more evapotranspiration means less soil moisture will be available for plant growth and less run-off, which is critical in regions with marginal rainfall (Houghton, 2009). Drylands where precipitation is low and where rainfall typically consists of erratic, short, high-intensity storms, cover c. 40% of total terrestrial land and support a fifth of the world's population. Desertification in these lands by climate change and human activities will lead to decreased vegetation and more soil erosion. The progress of desertification will be increased by land use and human needs (Houghton, 2009).

Table 3.1: Continued from previous page

Marine Protected Areas	
Risk	Impact on Protected Areas
Ocean acidification	When carbon dioxide dissolves in seawater, carbonic acid is formed, which then breaks down into bicarbonate and hydrogen ions. This leads to a decrease in the pH level and therefore an increase in the acidification level, which can have a significant effect on marine organisms that use calcium carbonate for their skeletal structures. This can degrade the structural integrity of coral systems as well as the skeletons of individual organisms (Speight and Henderson, 2010). A possible impact of ocean acidification on fish includes the behavioural effects on larval fish in sensory behaviour and detecting olfactory cues from habitats, and responding effectively to their surrounding environment. This can result in ecologically deleterious behaviour, affecting the survival of a number of fish species, and could eventually lead to biodiversity loss (Munday et al, 2010; Simpson et al, 2011).
Coral bleaching	For coral atolls, growth is inhibited by a sea temperature rise exceeding 1-2oC (Bijlsma, L. 1996). Coral bleaching has become a global phenomenon (Baker et al, 2008). It is caused by the expulsion of symbiotic zooxanthelae from coral. These symbionts are important for coral survival. Bleaching occurs when seawater temperature is elevated by 1-2oC for five to ten weeks during the summer season (Sampayo et al, 2008). Bleaching results in the death of the coral. Without a living coral reef system, the biodiversity of the ecosystem decreases. Habitat complexity will decrease, and therefore the species richness of fish and invertebrates due to a loss of a three-dimensional mosaic (Speight and Henderson, 2010). Dead coral that is not replaced with living coral will lead to eventual erosion and collapse of ecosystem complexity. Ocean acidification may also be involved in coral bleaching as well as temperature change (Anthony et al 2008).
Sea level rise	Melting polar ice caps and thermal expansion contribute to sea level rise. Coral atolls, such as the Maldives in the Indian Ocean, will suffer (Houghton, 2009). The global estimated average of sea level rise is 2mm per annum. Not only do half a million people live in archipelagos of small islands and coral atolls, but these areas of unique, endemic biodiversity could be wiped out completely, along with their human populations, including indigenous communities. Islands are known for accumulating unique biodiversity due to their isolation from terrestrial mainland. Coastal protected areas are particularly at risk. For coral atolls, rise in sea level at a rate of up to half a metre a century can be managed by coral growth, providing growth is not disturbed by human interference. (Bijlsma, L. 1996). Whether sessile species such as hard coral can grow sufficiently fast to compensate for their slow inundation is unknown.
Impact on mangroves and wetlands	Mangroves are a taxonomically diverse group of salt-tolerant, arboreal, flowering plants that grow primarily in tropical and sub- tropical regions (Ellison and Stoddart, 1991). Mangroves are vital to the world's ecology and total biodiversity. Wetland and mangrove swamps occupy about a million square kilometres of the Earth's surface and contain much biodiversity; their biological productivity exceeds that of any other natural or agricultural system. More than two thirds of fish caught for human consumption are dependent on mangroves for at least a part of their lifecycle, as do many other birds, mammals, reptiles and amphibians (Houghton, 2009). There is no evidence that mangroves can keep pace with a rate of sea level rise greater than 2mm per year, 20 cm per century (Houghton, 2009). Wetlands will extend inwards, but in many places extension will be limited by the presence of flood embankments and other human constructions, resulting in a loss of wetland area. Coastal wetlands are currently being lost at 0.5-1.5% per annum (Houghton,2009). Mangroves are less likely to suffer from temperature fluctuations, since they are well adapted to humid tropics and experience high temperatures (McCleod et al, 2006). Increased levels of CO2 are expected to enhance photosynthesis and mangrove growth rates (UNEP 1994). Although mangrove systems are known for being highly resilient, with climate change, this resilience may be threatened. Large storm impacts have resulted in the mass mortality of Caribbean mangrove forests in the last 50 years (Jimenez et al. 1985; Armentano et al. 1995). Degradation of coral reefs caused by mass bleaching and impaired growth damages coral reefs, which may adversely impact mangrove systems that depend on reefs to provide shelter from wave action (Hoegh-Guldberg 1999; McCleod et al, 2006). High water events could affect mangrove health and composition due to changes in salinity, recruitment, inundation, and changes in the wetland sediment budget and storm surges can flood mangroves (Gilman et al. 2006). Flooding, caused by increased precipitation, storms, or relative sea-level rise may result in decreased productivity, photosynthesis, and survival as well as destruction of mangroves (Ellison 2000).

The global mean sea level has experienced an annual increase, averaging 1-2mm per annum during the 20th century, with a further increase to about 3mm per annum from 1993-2003 (Lowe and Gregory, 2006; Parry et al, 2007; Bindoff et al, 2007). Rising sea levels promote land loss, increased flooding and salinisation (Nicholls and Mimura, 1998). The melting of the polar ice caps has been a well-recorded effect of climate change since the mid to late 20th century, which is the major contributor to rising sea levels (Chen et al, 2006; Christoffersen, 2006; Polyakov et al, 2010). The Arctic sea ice in particular has thinned by 40% in recent decades in the late summer to early autumn and has decreased in extent by 10-15% since the 1950s in spring and summer (Nicholls and Mimura, 1998; Christoffersen and Hambrey, 2006; Houghton, 2009).

Land possesses a much smaller thermal capacity and will respond more quickly to climatic changes. With warming at the earth surface there is increased evaporation from land areas as well as the ocean leading to an average increase in atmospheric water vapour content and therefore, on average, to increased precipitation. This further contributes to increased sea levels, as well as a reduction in freshwater sources and the melting of non-polar glaciers, snow cover and permafrost (Nicholls and Mimura, 1998; Parry et al, 2007; Houghton, 2009). The duration of ice cover on rivers, lakes and non-polar glaciers has decreased by about two weeks per annum over the 20th century, with the most drastic declines in ice cover duration seen in the mid and high latitudes of the northern hemisphere (Houghton, 2009).

Increased oceanic temperature is expected as well as increased ocean acidification due to the absorption of carbon dioxide by the ocean, converting CO2 into H+ and HCO3¬-¬¬¬¬, and decreasing the pH of the waters (Hoegh-Guldberg, 2007; Speight and Henderson, 2010). Ocean acidification compromises carbonate accretion, an essential function of coral formation, thereby decreasing the survival and reproduction of coral reef systems. Coral reef systems provide a three-dimensional ecosystem, an essential habitat for millions of species and vital for the maintenance of high levels of marine biodiversity (Hoegh-Guldberg, 2007; Speight and Henderson, 2010). Rising oceanic temperatures promote coral bleaching: the thermal tolerance of corals and their photosynthetic symbionts (zooxanthellae) is exceeded, resulting in the loss of the photosynthesising symbionts necessary for survival (Hoegh-Guldberg, 1990). Mass bleaching has already resulted in significant losses of live coral in many parts of the world. Like terrestrial forests, coral reef systems can often be centuries old and have a slow recovery rate once damaged extensively.

Many Marine Protected Areas (MPAs) are highly dependent on coral reef systems for the maintenance of high biodiversity levels, habitat diversity, and for tourism. Extensive coral reef structures often provide protection to coastal areas from incoming storms and heavy wave flow (Speight and Henderson, 2010). Marine protected areas, particularly reef systems, are in danger of a number of risks related to rising sea levels, ocean acidification, and increasing water temperatures (Speight and Henderson, 2010).

High altitude and mountainous PAs are likely to experience a loss of glaciers, snow cover and permafrost (Solomon et al, 2007). The result will be a loss of low-temperature-dependent species due to the lack of habitat. Particularly at risk are species incapable of migrating higher up a mountain or across a complex landscape (Primack, 2010; Marris, 2013). Lower altitude habitats dependent on melting ice and permafrost for freshwater may also suffer from increased drought and potential desertification.

In tropical and semi-tropical regions, rainfall patterns can be strongly influenced by the surface temperature of the oceans around the world. The Pacific Ocean off the coast of South America in particular is greatly affected every few years by the persistence of a large area of warmer waters (Houghton, 2009). This phenomenon, known as an El Niño event, causes warm ocean surface waters to prevent deep-sea nutrients from reaching the surface. Such nutrients are required by fish populations for survival, and these events have been known to devastate the fishing industry. Although this phenomenon has occurred for centuries, the advent of climate change increases the severity of El Niño events which reverberate on terrestrial lands and have consequences for PAs.

These include an increase in droughts and tropical forest fires in South East Asia and in the Amazon (Cochrane, 1999; Nepstad, 1999; Siegert et al, 2001; van der Werf et al, 2004). For example, during the 1997-98 El Niño, intense drought conditions enabled widespread fires across southeast Asian forests, Central and South American forests, and boreal regions of Eurasia and North America (van der Werf et al, 2004). As tropical forests are maladapted for fires, the impact is severe, the recovery rate is poor, and the forests are left at increased risk of future fires (Cochrane, 1999; Nepstad, 1999; Siegert et al, 2001; van der Werf et al, 2004). Forests may also be affected by the increased frequency of hurricanes, windstorms, ice storms or landslides that are associated with climate change (Dale et al, 2001).

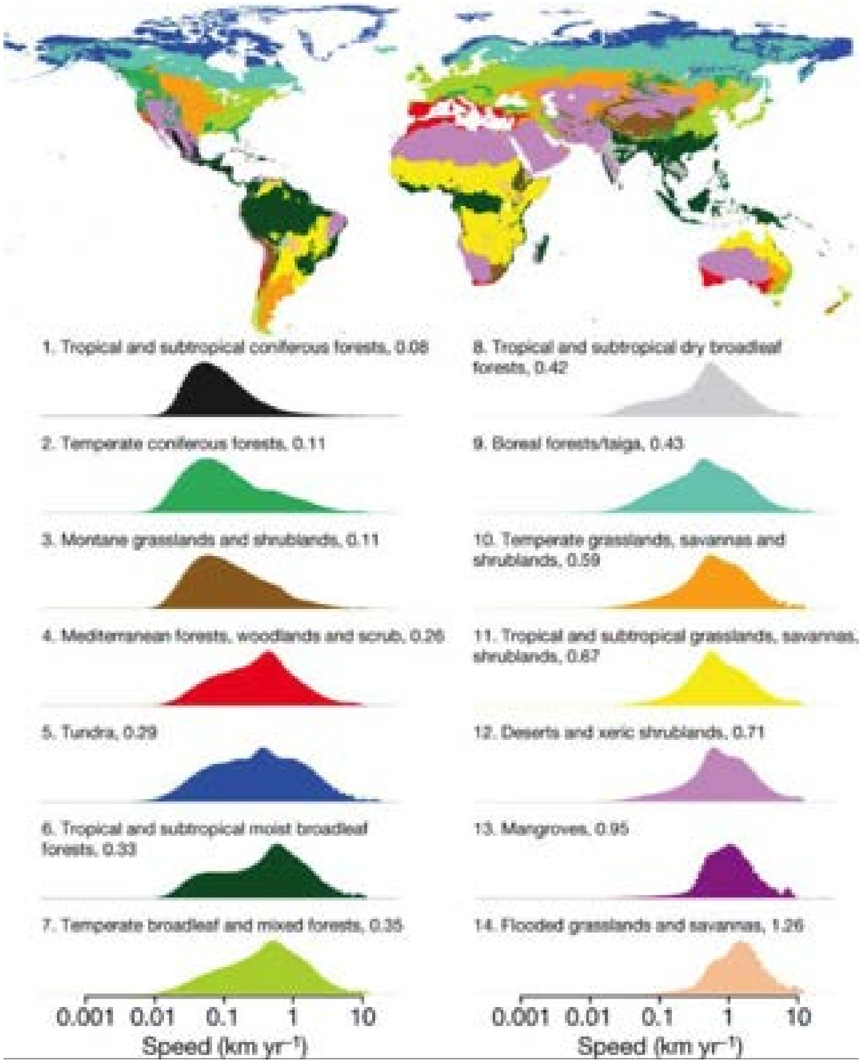
Risks affecting populations of species include loss of habitat, range shifts, changes in feeding and breeding pattern, an increase in invasive species (due to migration and competition in a changing climate), and possible increased occurrence of disease (Hannah, 2002; Hannah et al, 2007; Hellmann et al, 2008; Rahel and Olden, 2008; Dawson et al, 2011; Marris, 2013). These problems occur in both terrestrial and marine environments (Bax et al, 2003). Range shifts due to climate change may cause species to move out of protected areas. If nearby habitat is unavailable, it could lead to population decline, species extinction, competition with species in a new habitat, or encourage the growth of invasive populations (Hannah, 2007). Introduced species can affect habitats through predation, alteration of gene pools, and as disease vectors. They may also alter diversity, nutrient cycles, forest succession, and fire frequency and intensity for some ecosystems (Dale et al, 2001; Hansen et al, 2001).

3.1.2.3 Manageability

Each existing and new PA is likely to respond differently to the effects of climate change. Thus, PA management must cater for each unique biome. Due to the difficulty in creating accurate predictions for local and regional climate models, protected area management in the 21st century may need adaptive management strategies, approached as a continuing learning process involving trial and error responses to novel climatic events. From this, important lessons and data from real-time experiences can be shared across academic and professional platforms (Dale et al, 2001; Noss, 2001).

Collaborations across disciplines are recommended. Technological data from bioclimatic models, and Geographical Information Systems (GIS) are useful approaches for modelling landscapes for precipitation or disturbance, whereas population monitoring is useful for species data (e.g. Dale et al, 2001; Clark et al, 2001; Jeschke and Strayer, 2008; Aitken et al, 2008; Loarie et al, 2009). An understanding of species biology and local ecosystems is crucial for biodiversity conservation. An understanding of local animal ethology such as migratory patterns, breeding patterns and feeding preferences is beneficial for predicting the needs of adapting species (Dale et al, 2001; Festa-Bianche and Apollonio, 2003; Primack, 2010). A database on predicted species responses to climate change exists and continues to expand (e.g. Clark et al, 2001; Walther et al, 2002; Festa-Bianchet and Apollonio, 2003; Davis et al, 2005). Such data can be used to assist species adaptation, such as through assisted migration or captive breeding and seed banks when necessary (Leishman et al, 2000; Honnay et al, 2002; Ooi et al, 2009; Marris, 2013). This adaptive form of management can be used to determine how climate change affects populations and complex ecological interactions. Active elimination of invasive species to reduce competition when necessary is an example of species management that is already underway (e.g. Hannah et al, 2007; Rahel and Olden, 2008). Table 3.2 lists risks that protected areas are exposed to by climate change, and how these can be counteracted with management strategies.

Figure 3.2: *The velocity of temperature change by biome. A map of biomes and histograms of the speed of temperature change within each biome. Histograms are ordered by increasing velocity according to their geometric means*



Source: Scott et al., (2009)

The incidence of fire in the Amazon has increased substantially during the past decade (Cochrane et al, 1999). Accidental fires have affected roughly 50% of existing Amazonian forest, and have caused more deforestation than intentional clearing has in recent years (Cochrane et al, 1999). Current land use and fire use practices could in time transform large areas of tropical forest into scrub or savannah. Active monitoring to reduce accidental fires and prevent fire spreading, including the removal of ground fuel load, are advisable for forest management. Regular monitoring will also help to predict the incidence of fires and encourage preparation. Evidence suggests that selective logging in tropical rainforests may lead to an increased susceptibility of forests to fire as primary forests are less affected than recently logged forests (Siegert et al 2001). Fires severely damage forests by increasing the fuel load of dead flammable wood (Siegert et al, 2001). Minimising the occurrence of logging in old forests in PAs will help with reducing forest fires as well as maintaining a moist microclimate.

The tourism industry is beginning to face the profound impacts of climate change. Protected areas are a major source of tourism revenue, and tourism represents one of the world's largest industries, accounting for 9% of global GDP (Nicholls, 2014). Improving PA management for the benefit of tourism is important for value creation and for securing revenue. Sea level rise and ocean acidification can threaten coastal tourism infrastructure and natural attractions, and rising temperatures can shorten the period available for winter attractions such as skiing (Nicholls, 2014). A reduction in biodiversity can affect the potential for ecotourism (Viner and Agnew, 1999). Locations at risk could invest in more resilient infrastructure, or consider relocating. For example, about a third of Caribbean resorts are less than 1m above the watermark. A sea level rise of 1m would damage 49-60% of the region's tourist resort properties, the loss or damage of 21 airports, and inundate land around 35 ports (Nicholls, 2014).

Table 3.2: *Management strategies*

Risk	Management Strategy
Loss of land with sea level rise	Coastal PAs are those at imminent risk of sea level rise. Management strategies for these forms of PA: Physical barriers: coastal armouring and flood barriers. Suitable for PAs and coastal regions near urban centres and civilisation. A barrier, levee or sea wall could be fixed in place to allow managed flow through a portal for water exchange, tidal function and navigation. Barriers could alternatively be temporarily deployed to resist severe flooding during a storm surge. A downside is that such structures could be expensive and potentially ecologically damaging, and therefore more suitable alongside harbours or urban development. Hard shoreline protection is not as effective as natural shorelines at dissipating the energy from storms, waves and tides. Armoured shorelines tend to be more vulnerable to erosion and increase erosion of nearby beaches (SPUR, 2009). Wetlands, tidal marshes and mangroves, natural forms of shorelines, are adapted to absorb flood, waves, rising tide, slow erosion and provide habitat. Encouraging mangrove and wetland development is an ecologically effective way of supporting a PA and biodiversity and protecting coastal ecosystems from development impacts. Managed retreat of non-aquatic shorelines: planting trees or extending habitat further from the shore where possible, to allow organisms to move their habitat further inland in time before significant sea rise occurs and to accommodate natural coastal processes.

(Continued on next page)

***Table 3.2:** Continued from previous page*

Risk	Management Strategy
Tourism	Improving PA management for the benefit of tourism is important for value creation and for securing revenue. Locations at risk could invest in more resilient infrastructure, or consider relocating. Managed retreat strategies of near-shore terrestrial ecosystems could be applied to coastal resorts which depend upon coastal ecotourism for revenue generation. For example, about a third of Caribbean resorts are less than 1m above the watermark. A sea level rise of 1m would damage 49-60% of the region's tourist resort properties, the loss or damage of 21 airports, and inundate land around 35 ports (Nicholls, 2014). Snow-based resorts such as ski resorts may consider developing infrastructure to cater for more tourists during peak cold seasons where increased desiccation and snow-melt will occur during the remainder of the year.
Fire	Fire management planning should be based upon legal, institutional and policy frameworks. Adaptive management is also necessary as this revolves around the inconsistent nature of fire occurrence. Predictive monitoring: predicting fire occurrence is key to successful fire management. Data for pre-empting fires occurrence can come from satellite data, such as Moderate-resolution Imaging Spectroradiometer (MODIS) data. The Fire Information for Resource Management System (FIRMS) is another example. Studies have demonstrated the utility of remote sensing for monitoring Earth's surface and collecting data on fire occurrence (e.g. Kaufman et al, 1989; Barbosa et al 1997; Eva et al, 1998; Davies et al, 2009). MODIS data from NASA's satellites generate land surface data on the distribution of global fire occurrence, with FIRMS subsequently established to expand the distribution of MODIS fire data to a broader range of fire and forest monitoring organizations around the world (Davies et al, 2009). FIRMS is currently funded by NASA's Decision Support Programme and is working with the United Nations Food and Agriculture Organization (UNFAO) to establish a global fire monitoring capability for the United Nations (Davies et al, 2009). The application of this satellite use is targeted for accommodating a broader spectrum of users, including individuals with little to no remote sensing expertise, but who can benefit from access to satellite-derived fire information. Ground management: If a fire occurrence has been recognised in advance, there is a greater chance for ground staff to be prepared and conduct safety measures, including warning citizens as well as active and frequent management to reduce accidental fires and prevent fire spreading, such as the removal of ground fuel load. Reduction of logging, including selective logging methods, during fire seasons, even in tropical rainforests, may reduce fire susceptibility (e.g. Siegert et al, 2001). Related fire management strategies from the Amazon and Australia include Moran et al (2006 – Amazon); Moutinho et al (2009 – Amazon); Jurkis et al (2003 - Australia); The Institute of Foresters of Australia (2006 – Australia); Department of Parks and Wildlife Australia (Australia).

3.1.3 Conflict between communities and authorities

3.1.3.1 Introduction

Since the 1990s a group of social scientists have critiqued protected area policy and foregrounded the injustices wrought on local people. This critique, which is in coming under criticism for a its week historical approach, over generaisation and over-reliance on accounts in a limited literature, frames protected policy as having been been guided by an exclusionary approach, typically involving the forced removal of people from their homes and/or significantly curtailing their activities to minimise anthropogenic impacts (Lele et al, 2010). Integral to the first national parks established in the United States (Jacoby, 2003), this practice is seen as subsequently spread across the globe in diverse colonial settings, and was embraced by governments in the developing tropics after independence (Neumann, 1998).

The export of the US national park model approach has been criticised for ignoring other important social, cultural, and political issues. Brockington and Igoe (2006) document cases of complete physical displacement or eviction; economic displacement through restrictions on resource use (e.g., on collection of firewood and other non-timber forest products, grazing, and water use); and cultural displacement through restricted access to locations of cultural and symbolic value. These resulted in adverse social-economic impacts that disrupted traditional ways of living in local communities and restricted access and ownership to natural resources (Andrade and Rhodes, 2012). Although estimates of these displacements are highly uncertain, the alarming numbers ranging from 900,000 to 14 million people on the African continent (Geisler, 2003) and 100,000 to 600,000 in India (Lasgorceix & Kothari, 2009) have raised concerns. Forceful evictions and negative impacts on the lives of locals created much animosity, giving rise to conflicts between park managers and local communities (Brockington, 2002). Political ecologists have labelled such practice that forcefully excludes local people who have traditionally relied on and lived in the environment in question as 'fortress conservation' (Brockington, 2002).

The so called 'fortress conservation' approach has been dated back to the late 19th century in the creation of the world's first national park in 1872 (Kothari et al, 1995). The establishment of Yellowstone National Park in America involved the expulsion of Native American tribes via an ambiguous treaty (Brockington, 2004). What ensued gave rise to a series of escalating conflicts with park authorities and eventually violent clashes (Brockington, 2004). As colonial governments began to establish protected areas to conserve wildlife species of social and economic importance in Africa, the dominant fortress conservation approach was applied. Hunting practices by natives were presented as cruel, barbarous and wasteful (Adams and McShane 1996; Lewis et al. 1990; Neumann 1998) and conceptualised as a conservation problem. It has been estimated that over 70% of the population in Africa has been hurt by the conservation policies of colonial powers and independent governments, which include displacements exceeding 14 million (Veit and Benson, 2004).

Growing awareness of the presences and needs of local people in the 1970s began to encourage many protected area managements to incorporate the concept of participation within their management plan. During this period, there was a surge in the establishment of local support committees and communal reserves targeted at serving the needs of the locals (Stolton and Dudley, 2010). These initiatives, evaluated in the 1980s, as tensions between local communities and authorities continued to rise, revealed only limited engagement with locals and that inadequate authority was relinquished to the communities and locals when it came to decision-making (Stolton and Dudley, 2010).

Conflicts between local communities and PA authorities have complex roots and represent a major challenge in protected area managementHence, it is crucial to understand the ingredients of these conflicts in order to formulate sustainable management strategies in protected areas.

3.1.3.2 Materiality

Recent work by Kideghesh (2006) looking into the persistence of conflicts between authorities and local communities showed that they are fuelled by the inability to settle disputes over needs, perceptions, power and values. Neglecting these specific factors will only allow disputes to escalate into various forms of social unrest (see Section 3.1.11).

Local conflicts can be classified under four main headings: i) substantial; ii) psychological; iii) procedural and iv) ideological. Substantial and psychological conflicts, which are associated with needs and perceptions, are systemic and extend across protected areas, while procedural and ideological conflicts over processes, rights and claims are specific to individual or groups of protected areas in the same region. Table 3.3 sets out the various threats these conflicts present to PAs and their potential negative impacts.

Although initiatives to promote cooperation between local communities and authorities have been widely implemented in the 1970s and 80s, these have managed to address only the systemic issues, and not the specific conflicts that require complete integration of local views and actions in protected area management (Stolton and Dudley, 2010). This was recognised at the World Parks Congress in Durban in 2003, which called for an international agreement to fully assimilate local planning with conservation planning in protected areas (Hockings et al, 2004). Since then, the need to recognise all stakeholders as equal partners in global conservation efforts and harness indigenous and local knowledge (IDK), has been emphasised in recent protected areas and biodiversity governance policies – as indicated in the IPBES Conceptual Framework (Díaz et al, 2015) and the 2014 World Parks Congress in Sydney (Warne, 2015). There is a growing recognition of the role that traditional knowledge plays in increasing community resilience and capacity to mitigate and adapt to climate change, as reflected in the Fifth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC, 2013). These trends mark progress towards the assessment and analysis stage of the risk management cycle (Burgman, 2005). Hence, protected area management should make preparations for the development of management plans and the subsequent monitoring and control stages.

3.1.3.3 Manageability

A common feature of PAs located in developing countries is a funding deficit (Bruner et al, 2004). Andrade and Rhodes (2012) highlighted that by increasing long-term political and financial commitment in projects that enhance partnerships with local communities, a win-win outcome could be achieved. Funding which has been utilised to relocate and police protected areas can be channelled into improving governance, local capacity building, participation, and outreach programmes instead. Boissière et al (2009) have reported one such initiative in central Vietnam where successful cuts in patrolling and management costs allowed more to be spent on projects benefiting local communities, and significantly reduced both the degradation of forests and conflicts with park authorities.

Political and financial resources have been increasing over the past decades. This is evident in the recent work conducted through global and national platforms, such as the Sustainable Development Goals, the CBD, the UNFCCC, the UNCCD and the World Heritage Convention, which has begun to harness political support and build momentum. Meanwhile, funding interest has also risen in recent years. For instance, the Global Conservation Fund (GCF), a collaboration between Conservation International and the Gordon and Betty Moore Foundation has invested US$66 million in conservation projects that target local communities, generating more than US$1 million in wages for various local economies. More recently in 2015, the Global Environment Facility (GEF) announced a US$90 million grant programme aimed at encouraging local partnerships to promote both conservation and sustainable livelihoods of local communities in Africa and Asia.

Management in protected areas should take advantage of the current availability of political and financial resources and utilise them to review existing strategies and initiate surveys to understand the local community and their needs, perceptions, power and values. This will help to identify the specific types of conflict present, select appropriate management strategies and initiate partnership programmes. For instance, substantial conflicts that involve money, resources and time could be tackled by reviewing and developing policies that better support collective land rights. Some of the possible management strategies for the conflicts previously identified are detailed in Table 3.4 (above). Such a survey will also aid in the identification of required training for PA staff to build up their capacity in managing conflicts. The scale of these initiatives could be increased progressively according to the funds available and as international frameworks are formalised.

Table 3.3: *List of the type of conflicts, contributing factors, resultant threats and their potential negative impacts on protected areas (after Kideghesho, 2006).*

Type of conflict	Contributing factors	Threat	Potential Negative Impacts
Substantial	Money, resources and time	Inefficient allocation of financial resources	Downsizing of protected area
		Security and persistence of biological resources	Increased instances of illegal extraction, deforestation and poaching Degradation of biophysical values in protected areas
Psychological	Trust, fairness, participation and respect	Hinders the implementation of policies	Decreased competency in addressing impacts holistically
		Damages reputation of protected area and	Reduced donor and stakeholder support Affects brand value and potentially driving down visitorship
Procedural	Processes, traditional practices, rituals, protocols	Hinders the implementation of policies	Decreased competency in addressing impacts holistically
Ideological	Rights, claims	Ineffective conservation plans	Failure to meet conservation targets
		Social unrest	Protests, sabotage and disruption of protected area operations

__Table 3.4:__ Suggested management strategies to address the four types of conflicts identified

Type of conflict	Constituents	Possible management strategies
Substantial	Money, resources and time	Support collective land and resource rights
		Promotion of indigenous management of economies and special funding windows in existing mechanisms.
Psychological	Trust, fairness, participation and respect	Recognise the intrinsic and cultural values of local communities
		Increasing dialogues and participation of protected area staff in activities organised by local communities
Procedural	Processes, traditional practices, rituals, protocols	Co-creation of programmes with the full consent and involvement of traditional knowledge holders
		Applying traditional knowledge, practices and indigenous economies in the conservation and management of protected areas
Ideological	Rights, claims	Support collective land and resource rights
		Restoring rights of evicted communities and facilitating their return to and remain on their lands

3.1.4 Corruption

3.1.4.1 Introduction

Corruption is a major risk to protected areas as unlawful and exploitative management of PAs places all physical assets at risk and undermines the values of tourism, culture and biodiversity. Habitat loss from corruption exacerbates the risks of extinction for many species, while illegally seized timber or land destabilises the physical ecosystem structure.

Corruption is defined as 'the unlawful use of public office for private gain' (Transparency International, 2003) and is a widespread problem and systemic feature of many economies (Azfar et al, 2001). Political corruption involves individuals abusing their place in public office for financial or private gain (Smith et al, 2003). This often occurs because government officials in certain jurisdictions are poorly paid and susceptible to bribery. When such officials control valuable natural resources, whether oil, timber, minerals, gemstones or animal populations, these assets are at high risk of illegal extraction and trade (Kaufman, 1997; Laurance, 2004). Other factors that promote corruption include weak political institutions, poorly monitored resources, nepotism, and poorly updated checks and balances (Laurance, 2004).

Corruption has a highly corrosive effect on protected areas and on governments. It threatens the security of natural resources and therefore threatens economic development as well as social development, human welfare, animal welfare, biodiversity, and ecosystem services and functioning (Smith et al, 2003; Laurance, 2004).

Corruption can afflict all societies, from financial scandal and private enterprises defrauding investors in developed countries, to corruption in government in developing nations. Some countries however suffer from more corruption than others, as corruption appears to be linked to national economic welfare and GDP (Laurance, 2004).

3.1.4.2 Historical Context and Geographical Distribution

There is evidence that many developing countries, particularly those in the tropics with the largest proportion of global biodiversity, suffer from high levels of corruption (Smith et al, 2003; Wright et al, 2007). This is a major concern for the conservation value that PAs in these jurisdictions can generate (Myers et al, 2000; Cincotta et al, 2000; Laurance, 2004). Government officials who manage valuable resources such as oil, timber or minerals and who frequently engage in corrupt activities are probably the most serious risk to protected areas, especially those containing or proximate to natural resources.

Wealth in the form of natural resources can be detrimental to the economic development of a country (Koldstad and Wiig, 2009). Because corruption is a significant problem in developing countries that are rich in natural resources and biological diversity, this relationship is believed to be central in explaining why resource-rich countries perform poorly in terms of their socioeconomic development (Koldstad and Wiig, 2009). This phenomenon has been labelled the 'resource curse' or 'paradox of plenty' (Mehlum et al, 2006; Robinson et al, 2006; Koldstad and Wiig, 2009). The resource curse is the phenomenon of stagnant development and economic growth rates experienced by developing countries with an abundance of natural resources, fuels, minerals and biodiversity, in comparison to countries with fewer natural resources (Laurance, 2004; Smith and Walpole, 2005). Government corruption can often result from lack of stringent resource rights and unfair regulation of industry in an unequal society (Smith et al, 2003).

3.1.4.3 Materiality

Through unregulated management and exploitation, corruption is a significant risk to protected areas. For example, unlawful behaviour further degrades weak institutional structure, and unregulated exploitation of natural resources puts species at risk of extinction, degrades natural ecosystems and is often highly unsustainable.

Smith et al (2003) distinguish between non-collusive and collusive corruption. Non-collusive corruption increases costs for the private sector by imposing additional costs on business activity (Foellmi and Oechslin, 2007; Smith et al, 2003a). Governments demand bribes for a legal activity, such as obtaining a logging permit (Smith et al, 2003a). Non-collusive corruption is more widespread in low-income countries (Foellmi and Oechslin, 2007). Collusive corruption is a more decentralised form of corruption and it reduces costs for the bribee and tends to be more persistent within institutions. Collusive corruption can be more difficult to detect and eradicate, not only because it is decentralised but because neither the briber nor the bribee has an incentive to report or protest.

Individual government officials and the private sector scheme to collect revenues from what would otherwise end up in the government (Smith et al, 2003a). For example, individual government officials may allow exports without a permit, overlook tax evasions and allow the violation of sustainable logging operations or allow logging outside authorised areas (Smith et al, 2003a). Countries are particularly vulnerable to collusive corruption during political and institutional transitions because governments are unstable, underdeveloped and fragmented (Smith et al, 2003a).

Under President Suharto's regime in Indonesia, forests long used by local communities under informal rights, were declared state forests. Large-scale logging concessions were granted to forestry conglomerates controlled by Indonesian-Chinese entrepreneurs and government officials and military partners (Smith, 2003; McCarthy 2000; Barber and Talbott 2003). The military enforced obedience to Suharto's policies throughout the country, appointing officers to head provincial and district governments (Barber and Talbott, 2003). Illegal logging generates higher income for corrupt officials than legal logging, which further fuels unsustainable use (Palmer, 2001). Illegal logging and collusive corruption in Indonesia increased when the government was weakened after Suharto's regime and the responsibility for issuing permits was decentralised to local government.

Such institutional problems not only corrode governments and protected areas, they can devalue local culture by damaging the international reputation of the countries in question (Stapenhurst and Kpundeh, 1999). For nations such as Malaysia and Indonesia corruption in the natural resource extraction sector may detract from the positive perception of the relationship between the country's natural resources and society.

Some of the reasons why protected areas and conservation activities can be susceptible to corruption and poor governance include the low pay of officials, the external funding of projects, and the high value of many natural resources. Without fear of penalties and detection, illegal overexploitation continues to be a serious risk (Smith and Walpole, 2005). In the majority of national governments, the environment is still perceived as a 'soft' political issue (Smith, et al, 2003). In countries with weak governance, environmental goals can be overridden by extractive industries or construction, which can buy political influence (non-collusive corruption), reducing the stringency of environmental policies (Damania et al, 2003).

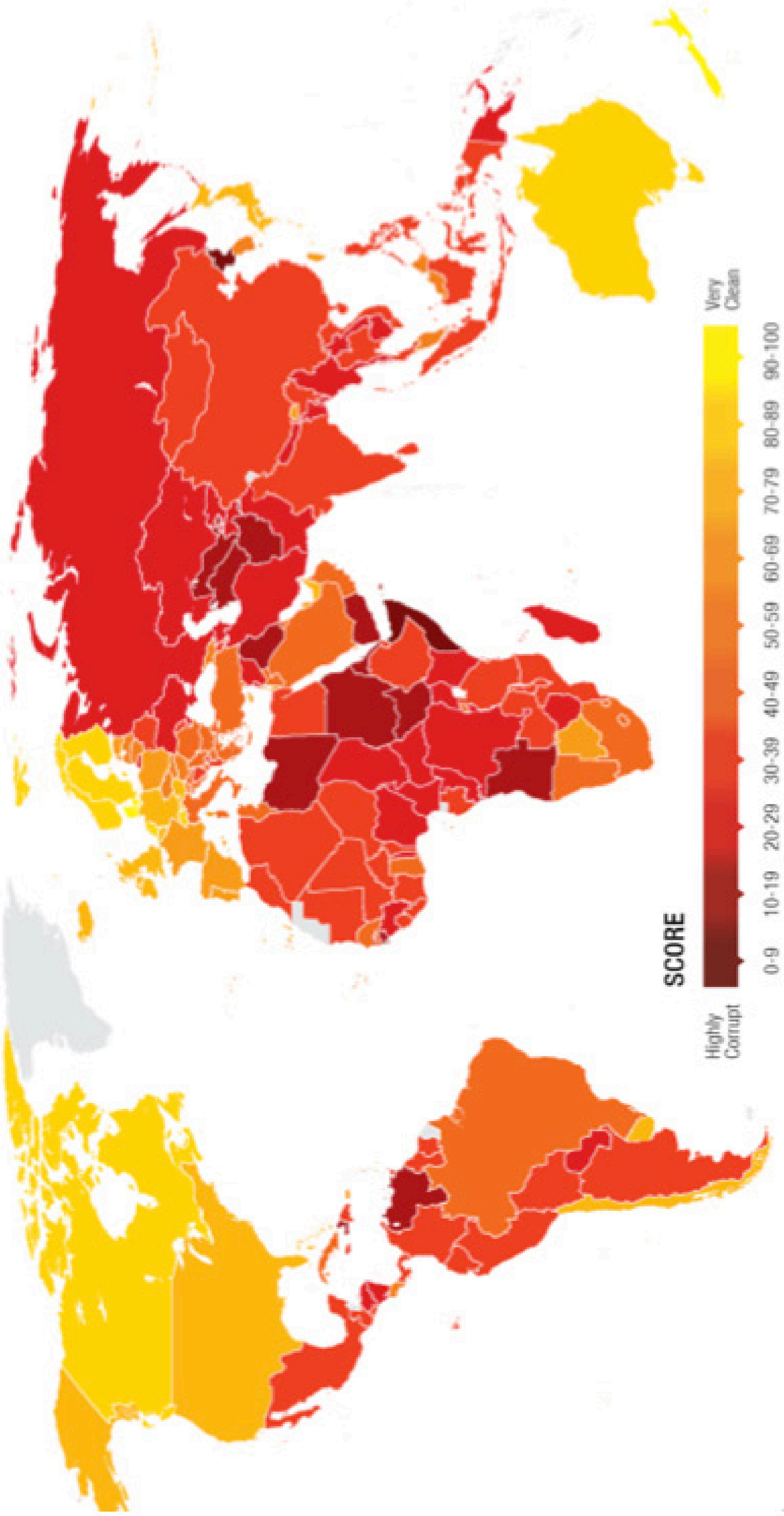
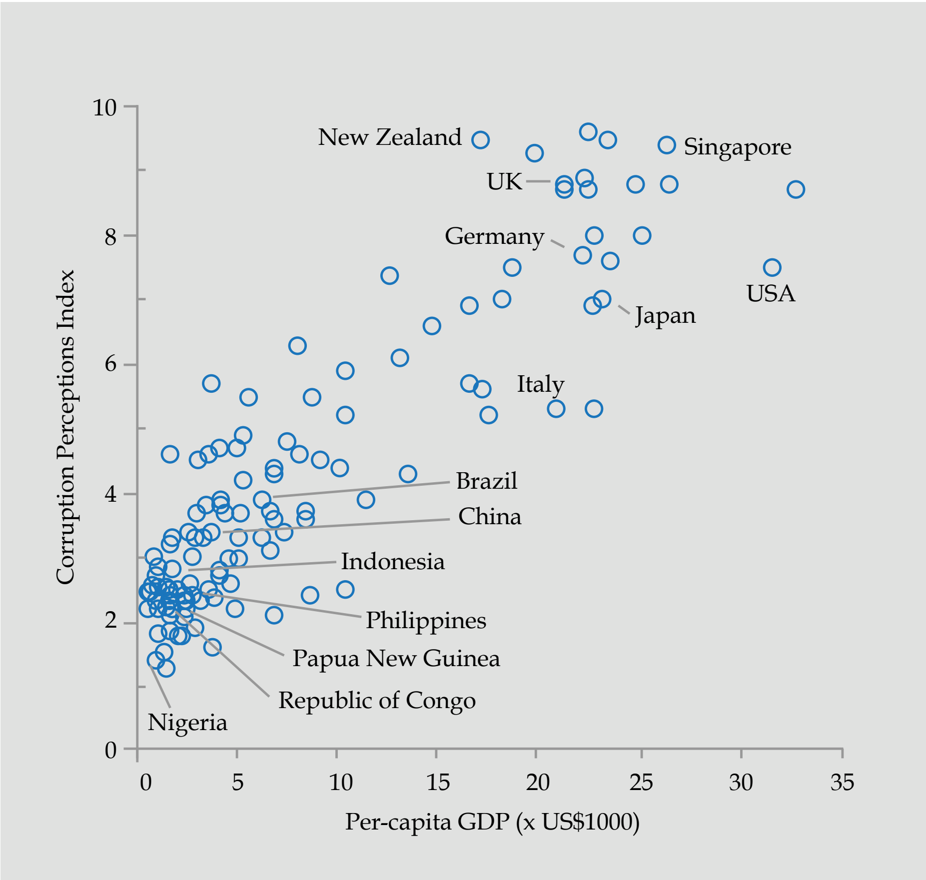


Figure 3.3: Corruption perceptions index scores 2014

Source: Transparency International, 2014 (http://www.transparency.org/cpi2014/results)

Figure 3.4: *The prevalence of corruption across nations. Shown is the relationship between per capita gross domestic product (GDP), which is an index of economic development and Transparency International's Corruption Perceptions Index in which lower scores indicate increasingly pervasive corruption. The relationship for 130 countries is highly significant (r=0.893, P<0.0001; Pearson correlation)*



Box 3.1: *Corruption in Indonesia*

Indonesia is an example of a country where corruption levels are high (Henderson & Kuncoro, 2004). Much of this corruption affects biodiversity and protected areas in the form of illegal wildlife trade, illegal logging and illegal deforestation (Callister 1999; Palmer 2001; Contreras-Hermosilla 2001; Scotland et al 2000; Smith et al 2003). For example, the Ministry of Forestry in Indonesia estimates $3.7 billion is lost annually due to illegal logging and exports (Smith et al, 2003). Bribes by firms in Indonesia arise principally from regulations, licences and levies imposed by local government officials. Such regulations can generate direct revenues and indirect revenues through bribes (Henderson & Kuncoro, 2004). Bribery becomes a significant source of personal finance. Bribes are used to compensate public officials and provide competitive salaries (Henderson & Kuncoro, 2004). Data from interviews in Indonesia show illegal logging became widespread after the fall of President Suharto (Smith et al 2003). During political transitions, governments are often weak and fragmented with underdeveloped institutions, making countries particularly vulnerable to corruption (Smith et al, 2003). To combat illegal logging and corruption in Indonesia government accountability must increase, and legal and judicial reform are necessary (Smith et al, 2003).

Source: Laurance (2004)

Corruption is considered to be so endemic in the multibillion dollar tropical timber industry (Palmer, 2001; Laurance, 2004; Lang, 2014) that the relationship has been described as symbiotic (Smith, 2003; Lang, 2014). Case studies suggest corruption could be a key factor in the loss of tropical forests through unsustainable or illegal logging (Huber, 2001; Jepson et al, 2001; McCarthy, 2002; Smith and Walpole, 2005). In 1996 illegal logging cost Indonesia US$660 million or 1.5% of its GDP (Palmer, 2001). World Resources Institute estimates have suggested that between 20-80% of all harvested timber may be illegally sourced (WRI, 2003). Furthermore, the physical impact illegal logging can have upon protected areas includes breaching ecological thresholds by degrading soil composition (soil erosion), pollution, increasing wild fire risk by adding fuel load (Cochrane et al 2002; Nepstad et al 1999), contributing to climate change, and disrupting animal populations (e.g. Plumptre et al, 1994), damaging biodiversity, increased edge effects and encouraging loss of topsoil and decreasing water retention through change in microclimate (Asdak et al, 1998; Broadbent et al, 2008).

In Uganda, corruption was viewed to be one of four main problems in revenue-sharing programmes for protected areas (Archibald and Naughton-Treves, 2001). For example, revenues generated from tourism in and around the Masai Mara National Reserve in Kenya were almost entirely misappropriated by local elites (Thompson and Homewood, 2002). Only 6.5% of funding intended for local community projects was dispersed, and most of it went instead to various administrative mechanisms alongside some direct embezzlement (Thompson and Homewood, 2002). Thus, many such projects can fail to reinvest revenues in the biodiversity conservation on which local communities rely.

3.1.4.4 Interrelationships

Corruption exists within governments and extractive industries, and therefore is related to conflicts between communities and authorities (see Section 3.1.3) and extractive industries (see Section 3.1.5). Corruption interlinks closely with the illegal wildlife trade, and has direct and indirect effects upon ecosystems and potentially breaches ecological thresholds, and is highly correlated with political instability.

3.1.4.5 Manageability

Even though conservation researchers and practitioners recognise corruption as a threat to biodiversity and seek to increase transparency and accountability (Smith and Walpole, 2005), the nature and magnitude of its impact on PAs is not fully understood. While precautionary measures can be included in PA management plans to account for the occurrence of corruption, further research is needed to develop more appropriate responses.

The role of corruption in national and international economies has been analysed by numerous theoretical and empirical studies (Smith and Walpole, 2005), however, few studies have empirically analysed the relationship between corruption and conservation. Therefore there is insufficient literature available to provide a reliable guide for conservation practitioners (Ferrero, 2005).

Corruption on a small scale in remote areas receives little scrutiny from the international – or even national – community, and is often difficult for outsiders to discover or publicise (Smith and Walpole 2005). For example, Brazil holds national and regional anti-bribery events to improve awareness in the private sector and civil society (OECD, 2003). Brazil also hosted the 2005 Global Forum on Fighting Corruption (OECD, 2003; Laurance, 2004).

By understanding the system of corruption, its culture and networks, it can be seen as not just the absence of rules, but also as the presence of alternative norms. Robbins et al (2000) consider that corruption exists in the presence of different institutions, not just the absence of state institutions, and is a culture in itself of 'trust, betrayal, deception, subordination of common to specific interests, secrecy, involvement of several parties, mutual benefits (material or pecuniary)' (Perry, 1997). The establishment of a corruption culture prevents conflicts with other morals or rationalities (Robbins, 2000).

Transparency is increasingly viewed as a prime means of mitigating corruption in resource rich developing countries¬ (Kolstad and Wiig, 2009). Initiatives such as the Extractive Industries Transparency Initiative (EITI) have been created by the international development community to encourage transparency in supply chains and resource revenues. Transparency International and the International Chamber of Commerce, are helping to promote intolerance of bribery, and educational activities on corruption topics in many countries, (OECD, 2003; Laurance, 2003). The involvement of multinational corporations in improving codes of conduct and public transparency could be beneficial in reducing non-collusive corruption. However, the role of transparency in understanding and reducing corruption and averting the resource curse is poorly understood and therefore may be insufficient by itself, and should be complemented by other types of policies (Kolstad and Wiig, 2009).

When surveillance is low and the price of a penalty if caught is insignificant, bribery and illegal extraction are likely to continue unabated. Especially in the case of collusive corruption, neither the briber nor bribee has sufficient incentive to report or protest. Anti-bribery campaigns are more meaningful with improved and enforcement. Chile and Korea, for example, have task forces that specialise in the prevention, detection and prosecution of acts of corruption and bribery (OECD, 2003; Laurance, 2004).

3.1.5 Extractive Industries

3.1.5.1 Introduction

Extractive industries have long been labelled as the primary cause of protected area degradation and destruction. The search for energy sources such as coal, oil and gas has been well recognised as a serious risk to protected areas. Non-extractive industries such as timber felling and land clearing for agriculture also have devastating impacts. The demand for profit spurs on the unsustainable 'business-as-usual' approach to resource extraction, and those working within the conservation sector know only too well its impact on protected areas. This report looks into these more familiar details of how extractive industries have affected protected areas in the past, and highlights the main risks that extractive industries currently pose to protected areas, the improvements some industries have made, and how the relationship between extractive industries and protected areas can be better managed to promote more sustainable forms of resource utilisation.

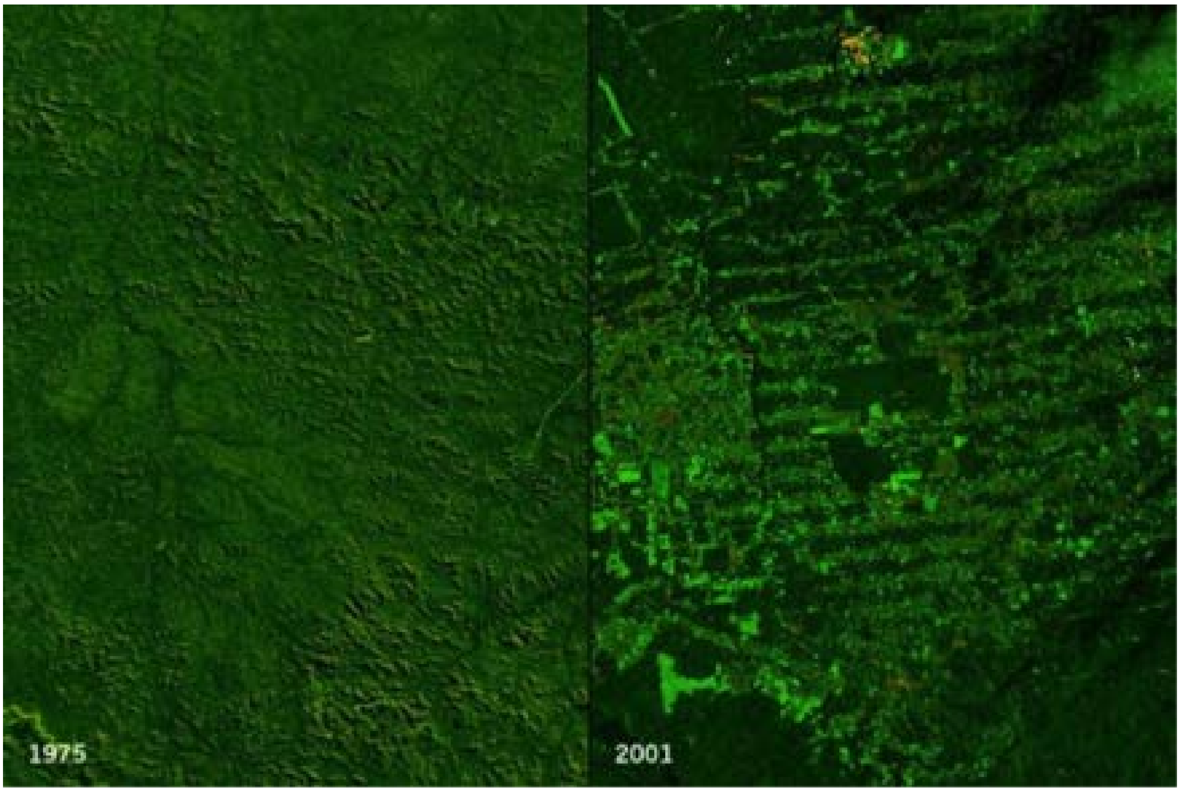
Extractive industries are concerned with the extraction of physical natural resources, specifically metals, minerals, oil, gas, coal, hard rock, sand, gravel and other aggregates. These raw materials are used in production for markets and by consumers. Demand is increasing for these materials in response to population growth, urbanisation, expansion in agriculture and industry, and rising global consumption. The process of extraction (e.g. mining) includes exploration, exploitation, extraction transportation, and processing, which have damaging impacts upon biodiversity, the physical land and ocean floors that support biodiversity, and other wider natural, cultural and economic implications. Values that protected areas safeguard are put at risk if extractive industries explore and transport raw materials from these sites (World Parks Congress, 2003).

3.1.5.2 Historical context and geographical distribution

Industrial extraction involves a number of high-risk activities along the chain of activities from exploration to transportation, and these affect habitat and species conservation as changes in land use directly affect ecosystems. Unsustainable mining generates pollution which can threaten plant growth and freshwater and food supplies, including drinking water for nearby human and animal populations (WWF, 2015). Human settlements along roads newly built for the purposes of extraction, in previously virgin or untouched ecosystems, carry a collection of new threats for the surrounding environment, including poaching and illegal encroachment upon protected areas (WWF, 2015). In many cases, extractive enterprises overlap with vulnerable indigenous territories, placing local and indigenous knowledge at risk (Suárez et al, 2009, Jobin, 2003). The absence or weakness of local governments, the lack of alternative forms of enterprise and revenue, and the often minimal knowledge about the effects of extraction can extend beyond ecological thresholds, drastically changing the social circumstances and livelihoods of local communities, and the wildlife resources upon which they depend (Suárez et al, 2009, Jobin, 2003).

The Amazon rainforest, for example, has been dramatically changed through a combination of activities from extractive and non-extractive industries (especially timber harvesting, legal and illegal) as satellite images show (Figures 3.5 and 3.6). Other large rainforest areas, particularly in southeast Asia and Central Africa show similar scenarios.

Figure 3.5: *Satellite images from 1975 (left) and 2001 (right) illustrating the reduction in Amazonian rainforest cover due to logging*



Source: Imazon (2013) and (Huguet, 2014)

Marine Protected Areas (MPAs) are marine biomes specifically designated for the protection of marine biodiversity, ecosystems and cultural resources. MPAs are less abundant than terrestrial protected areas (UNEP-WCMC, 2015), but stand at similarly high risk of ecological damage from extractive industries. The risk of marine oil spills is high, but can be reduced by transporting oil products in double-hulled tankers (Carter, 2005). Not all companies apply such practices at their facilities, however, and damage continues through poorly managed operations, in companies of varying sizes, and some of which operate illegally in protected areas (Carter, 2005). The designation of MPAs to limit the distribution of extractive, destructive and polluting activities has been embraced as a powerful tool for conserving marine ecosystems (Lubchenco et al, 2003; Game et al, 2009).

As traditional oil-producing regions progressively yield decreasing quantities of oil or other raw materials, extractive industries explore new areas and new products, such as natural gas (Carter, 2005). Exploration, extraction and transportation has been taking place, or is planned, in regions known for their rich biodiversity, including the Caspian, Indonesia, West Africa, offshore Venezuela and Trinidad (Carter, 2005). Maritime transport and pipelines are required to transport oil and liquefied natural gas to consumers across nations, and this requires the construction of pipelines that will cut across wilderness and conflict with both terrestrial and marine protected areas (Carter, 2005; Jaffe and Victor, 2004).

Figure 3.6: *Map indicating deforestation and forest degradation in the Amazon Biome*

Source: (Tollefson, 2013)

The economic significance of the extractive sector to producer countries is high. The sector also has a common role in influencing the fate of political leaders. As a result, extractives industries are often subject to intense global scrutiny, in the form of investigating revenue transparency and their environmental legacy or stewardship role (Stevens et al, 2013). The sector influences both national economies and local communities and remains an area of 'contested rights, responsibilities and benefits' (Stevens et al, 2013). High prices over the last decade, and increasing global manufacturer demand for raw extractive materials brought about a generation of 'mega-investments' (Stevens et al, 2013) in mineral extraction, in countries with long-established extractive industrial sectors (e.g. Australia, Chile, and Canada); the 'emerging producers' (Stevens et al, 2013) (e.g. Mozambique and Mongolia) attracted investments from private corporations or state-owned enterprises (SOEs).

The long-term future for extractive industries, in developed and developing countries, lies increasingly in geologically, ecologically and politically challenging regions. Issues such as water scarcity and concerns over resource security, environmental degradation or climate change will bring further scrutiny and tension to protected areas (Stevens et al, 2013). In many countries, conflicts with extractive industries are likely to escalate, which may have ramifications for the economic and political stability of nations, polities, company assets and reputations (Stevens et al, 2013).

3.1.5.3 Materiality

The global demand for oil, gas, minerals and metals is expected to increase (IEA, 2015). To supply global refineries and smelters, extractive companies may need to intensify prospecting and production efforts. Exploration in remote and virgin areas, many of which are already protected areas or candidates for protection, may continue or accelerate (Carter, 2005). While the recent fall in global commodity prices may dampen natural resource exploration , it is not yet clear whether this is a structural change in commodity markets or simply a temporary (albeit significant) downturn in activity (Krauss and Reed, 2015). This will probably reduce the pressure on natural resource exploration and extraction within or near PAs, but this may very well be shortlived.

3.1.5.4 Manageability

Technologies and management techniques for mitigating many of the impacts of mining and oil and gas development are already available (Carter, 2005) but are rarely employed (Heike et al, 2011). These techniques include replacing roads that cut through forests and other protected areas to extractive project sites with other modes of transport, such as helicopter transport; cutting seismic lines to less than 4m in width; burying oil pipelines underground; establishing canopy bridges across rights of way; replanting and restoration directly after extraction and ecological disturbance; preventing pollution for wildlife and local community safety; increasing investment for safety and pollution prevention measures instead of using this as money-saving area; and reducing the risk of marine oil spills by transportation of oil products in double-hulled tankers (Carter, 2005). However, under 'business as usual' approaches that advocate money-saving strategies at various points along the production chain, there is often resistance to adopt effective, ecologically sound management, which demands increased investment in production. PA management should adopt strict requirements to ensure that extractive industries employ the relevant technologies and management techniques to mitigate potential impacts prior to exploration and extractive activities. In addition, PA management should work together with extractive companies to identify specific 'no-go' areas (World Heritage, 2014).

Recent years have also seen an increasing number of major companies in the extractive industry adopting corporate social responsibility (CSR) models of business in order to improve their public image, promote their brand value, and attempt to reduce their impact upon the natural resources they extract, as well as their wider environmental impacts, such as contribution to climate change (Hilson, 2012; Slack, 2012). PA management should encourage such practices by recognising the efforts of these businesses via sharing of their initiatives with the wider public and conservation community.

Finally, PA management can also work with business to spur further investment in technology and management, particularly in the scientific exploration of increasingly geologically complex areas. Such partnerships between PA managers and the tourism industry may contribute to sustainability as the information gathered on species and various other features of protected areas provided results of importance to biodiversity conservation (Pfueller et al, 2011). This is especially the case in MPAs, potential MPAs and other marine biomes, which are targets of oil and gas exploration as well as important sites for biodiversity conservation.

Footnotes:

Recent falls in oil prices due to slowing demand and a glut of supply has led to the exit of Royal Dutch Shell from Arctic oil exploration (Krauss and Reed, 2015).

3.1.6 Geopolitical Conflicts

3.1.5.4 Introduction

Geopolitical risks include i) interstate conflict with regional consequences; ii) state collapse or crisis; iii) terrorist attacks; and iv) weapons of mass destruction, and are often linked to natural resources (Bavinck et al, 2013). The recent Global Risks Report published by the World Economic Forum (2015) identified geopolitical risks as the biggest threat to global stability in the next decade.

Since 1945 there have been approximately 30 cases of geopolitical conflict; of these Egypt in 1978, Ethiopia in 1991 and Rwanda in 1997 have specific investigations into the impact on protected areas (See Table 3.5). Other notable conflicts that implicated ecology and the environment, as outlined in a general study on biodiversity hotspots (Hanson et al, 2009), are shown in Table 3.6 and include the Sino-Indian conflict, the Vietnam War, the Iraq War (Alsdirawi, 1991) and the Gulf War (Al Houty et al, 1993). They resulted in huge destruction and degradation of the environment during and after the conflicts (Hanson et al, 2009).

Such conflicts can impact protected areas in different ways. There can be direct impact such as the aerial application of Agent Orange and other herbicides in the Vietnam War, which defoliated 14% of the country's forest cover and over 50% of its coastal mangroves (SIPRI, 1976; Hastings, 2000). Various cases of direct impacts such as habitat destruction and damage to infrastructure supporting conservation have also been identified in the essay by Dudley et al (2002). Apart from direct impacts, indirect impacts brought about by increased settlement in protected areas may lead to increases in pollution and bushmeat hunting (Hanson et al, 2009) and even affect protected areas far away from where the war is being waged. For instance, spending cuts were made to the US Forest Service in order to channel monetary resources to the war effort in Afghanistan, leading to the abandonment of various planned conservation projects (Daly, 2008).

Finally, the desolate state post-conflict countries are often left in also poses many challenges given competing priorities. Most post-conflict environments are characterised by large movements of people and increased competition for scarce resources in the search for sustainable livelihoods. Instances of illegal poaching, fishing, logging, and mineral extraction for subsistence or business purposes have been shown to increase post-conflict and has led to downsizing and degazettement of protected areas such as Parc National de l'Akagera in Rwanda (Mascia and Pailler, 2011). Hence, there certainly is a need to recognise these threats and continue to tackle them even after the resolution of geopolitical conflicts.

Considering the increasing global threat geopolitical conflicts pose, and the magnitude of negative impacts they constitute, better understanding of the management of risk is warranted to ensure that values in protected areas are not impaired.

Table 3.5: *Sources recording impacts of major geopolitical conflicts on protected areas*

Country	Protected Areas	Year	Source
Egypt	Ras Muhammed National Park	1978	World Wildlife Fund, 2014
	St. Catherine Mountain Reserve		
Ethiopia	Lake Mburo National Park	1991	Hillman, 1993a,b; Jacobs and Schloeder, 1993
	Gorilla National Park		
	Semliki Forest Reserve		
	Parc National des Volcans		
	Parc National des Volcans		
	Gishwati National Park		
	Mukura National Park		
	Kahuzi-Beiga National Park		
	Tigris-Euphrates Marshlands		
	Annapurna Conservation Area		
	Gola Forest Reserve		
	Luiana Protected Reserve		
Rwanda	Akagera National Park	1997	World Wildlife Fund, 2014
	Mutara Hunting Zone		

3.1.6.2 Geographical distribution

The 2014 Global Peace Index (GPI) produced by the Institute for Economics and Peace is an attempt to measure the relative position of nations' and regions' peacefulness and could be inverted to act as a simple indicator for geopolitical risk. Countries with a peacefulness index of less than 2.5 are hence more threatened by geopolitical risk, countries with a high GPI are less prone. Geographical distribution of protected areas in localities with high geopolitical risk is obtained by combining data on protected area size and distribution data obtained from the UNEP-WCMC World Database for Protected Area with the GPI.

This shows that 11.8% of the world's protected area is located in the 25 countries with high geopolitical risk (GPI < 2.5). More specifically, the protected areas in the Central African Republic, South Sudan, Egypt, Ethiopia, Chad, India, Colombia, Democratic Republic of the Congo and Russia accounted for more than 10% of the world's parks (Figure 3.7). The majority of these have experienced recent local or geopolitical conflicts and may face a high risk of resurgence due to sustained political instability.

Transboundary Protected Areas (TBPAs), owing to their location near the edges of geopolitical units (Goodale et al, 2003) and their function as key mechanisms to control and conserve natural resources are often implicated in conflicts (McNeely, 2005). A review conducted by Rustad and Binningsbø (2012) reported that from 1970 to 2006, almost 40% of all interstate conflicts are linked to high-value natural resources, with over 80% of all major armed conflicts between 1950 and 2000 documented within areas recognised as global biodiversity hotspots (Hanson et al, 2009). Looking across regions, almost half of the world's TBPAs overlap with the regions with high geopolitical risks in East Africa, across Central Asia and towards South Asia. Necessary precautions, resilience building and management protocols should be considered in these TBPAs to mitigate their high geopolitical risks.

Despite our best efforts to identify areas at high risk, the ability of climate change to overstretch adaptive capacities could still result in destabilisation and violence, even in countries with potentially low geopolitical risks (Haldén, 2007). The German Advisory Council on Global Change, (Wissenschaftlicher Beirat der Bundesregierung Globale Umweltveränderungen, WBGU), argues that climate change will lead to an increase in geopolitical conflicts in weak and fragile states, newly industrialising countries and in locales unable to manage the greater incidence of disasters (Schubert and Schellnhuber, 2009). The 2014 Quadrennial Defence Review, issued by the US Department of Defense shares the same sentiments and boldly mentioned climate change as a 'threat multiplier', emphasising the integral role climate change plays in determining geopolitical conflicts (Hagel, 2014). Currently, there is no published literature considering the synergistic effect climate change has on geopolitical conflicts within protected areas and future studies should aim at filling this gap for a more precise understanding of geopolitical risks relating to protected areas.

3.1.6.3 Materiality

The main objectives of protected areas are to support the 'long term conservation of nature with associated ecosystem services and cultural values' (Dudley, 2008). Although restrictions on human access in conflict and demilitarised zones has in some cases allowed vegetation and wildlife to flourish, such effects are highly variable (Austin and Bruch, 2000; McNeely, 1998). Positive impacts, if any, are often overshadowed by threats to the economy and development, efficacy of policies and security (Jacobs and Schloeder, 2001).

Figure 3.8 modified from Jacobs and Schloeder's (2001) analysis summarises the dysfunctions brought about by the breakdown in rule of law, increased abundance of firearms, disruption of economic activity, agricultural production and trade, increased dependence on wild resources and mass movements of people, and details the pathways leading to negative impacts like habitat destruction, over-exploitation of natural resources, and pollution in protected areas (Oglethorpe et al, 2004). Furthermore, Oglethorpe et al (2004) pointed out that theft, permanent damage or destruction to infrastructure, vehicles and equipment are frequent in protected areas during times of conflict, and disrupt management and surveillance programmes.

SMITH SCHOOL OF ENTERPRISE AND THE ENVIRONMENT

UNIVERSITY OF OXFORD

Table 3.6: List of conflicts and recorded in biodiversity hotspots from 1960 to 2008

Biodiversity hotspot	Nations represented in hotspot	Geopolitical conflicts (dates)
Atlantic Forest	Argentina, Brazil, Paraguay, Uruguay	Argentina-Montoneros Conflict (1973-77), Argentina-Dirty War Conflict (1976-1980)
Cape Floristic Region	South Africa	South Africa–ANC–PAC Conflicts (1981-1993)
Caucasus	Armenia, Azerbaijan, Georgia, Iran, Russia, Turkey	USSR–Ukraine–UPA Conflict (1945-1950), Russo–Hungarian War (1956), Iran–Mujahedin Conflict (1981-82), Georgia–Gamsakurdia Conflict (1991-94), Azerbaijan–Karabakh Conflict (1991-94), Russia–Chechnya Conflict (1992-2000)
Eastern Afromontane	Burundi, the Democratic Republic of Congo (DR Congo), Eritrea, Ethiopia, Kenya, Malawi, Mozambique, Rwanda, Saudi Arabia, Somalia, Sudan, Tanzania, Uganda, Yemen, Zambia, Zimbabwe	British–Mau–Mau Conflict (1952-56), DR Congo Conflicts (1960-65; 1993; 1996-97; 1998-present),Rwanda–Tutsi–Hutu Conflicts (1962-63; 1990-93;1994), Uganda–Baganda Conflict (1966), Burundi–Tutsi–Hutu Conflicts (1972; 1988) 1991; 1993-present), Uganda–Tanzanian War (1978-79), Ethiopia–Tigray Conflict (1978-1991), Ethiopia–Eritrea War (1998-2000)
Himalayas	Himalayas	India–Pakistan Kashmir Wars (1964-65; 1971; 1999), Sino-Tibetan War (1950-51), China–Tibetan Conflict (1956-59), Sino-Indian War (1962), Pakistan–Baluchi Conflict (1973-77), India–Kashmiri Conflict (1985–present), Pakistan–Mohajir Conflict (1994-95)
Horn of Africa	Djibouti, Eritrea, Ethiopia, Kenya, Oman, Saudi Arabia, Somalia, Sudan, Yemen	Kenya–Somalia–Shifta War (1963-67), Ethiopia–Eritrean Conflict (1974-1991), Ethiopia–Somalia Conflict (1976-77), Ethiopia-Somalia War (1977), Ethiopia–Eritrea War (1998-2000)

***Table 3.6:** Continued from previous page*

Biodiversity hotspot	Nations represented in hotspot	Geopolitical conflicts (dates)
Indo-Burma	Indo-Burma, the Andaman Islands (India), Bangladesh, Cambodia, China, Laos, Malaysia, Myanmar, Vietnam	Vietnam War (1965-1975), Pakistan–Bengali Conflict (1971), Bangladesh War (1971), Vietnam–Cambodia War (1975-79), Sino-Vietnamese Wars (1979; 1987)
Irano-Anatolian	Armenia, Azerbaijan, Georgia, Iraq, Iran, Turkey, Turkmenistan	Iraq–Kurd Conflicts (1961-63; 1974-75; 1985-1993; 1996), Iran-Iraq War (1980-88), First Gulf War (1990-1991)
Maputaland–Pondoland–Albany	Mozambique, South Africa, Swaziland	Africa–ANC–PAC Conflicts (1981-1993)
Mediterranean Basin	Albania, Algeria, Azores, Bosnia and Herzegovina, Bulgaria, the Canary Islands, Cape Verde Islands, Croatia, Cyprus, Egypt, France, Gibraltar (U.K.), Greece, Israel, Italy, Jordan, Lebanon, Libya, Macedonia, Madeira Islands, Malta, Monaco, Montenegro, Morocco, Portugal, Selvages, Slovenia, Spain, Syria, Tunisia, Turkey	Franco-Tunisian Conflict (1952-54), Israel–Palestinian Conflict (1955-present, Israel–Egypt War (1969-1970), Jordan–Palestinian Conflict (1970), Yom Kippur War (1973), Turko-Cypriot War (1974), Yugoslavia–Serbia–Croatia Conflict (1991-92), Bosnia and Herzegovina–Serb Conflict (1992-95)
Tropical Andes	Argentina, Bolivia, Chile, Colombia, Ecuador, Peru, Venezuela	Colombia–Violencia Conflict (1949-1962), Argentina–Montoneros Conflict (1973-77), Argentina-Dirty War Conflict (1976-1980)

Source: Selected geopolitical conflicts in Hanson et al, (2009) based on conflict data from conflict data from Arnold (1991), Sarkees (2000) and Gleditsch et al, (2002)

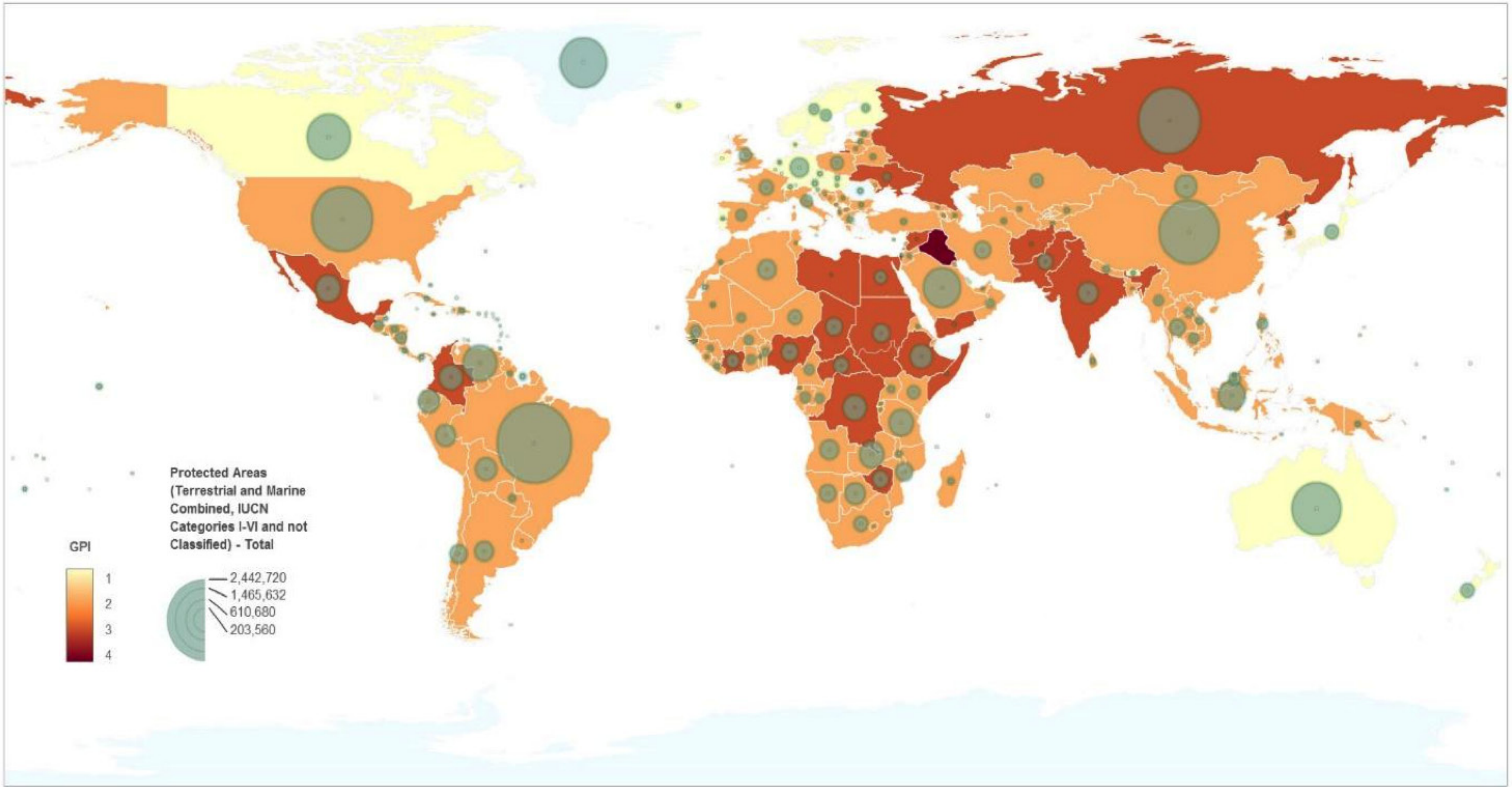


Figure 3.7: *Location of global protected areas (WDPA, 2015) within geopolitical risk map indicated by relations with neighbouring countries (World Peace Index, 2014)*

3.1.6.4 Manageability: what PA management can do to manage these risks

Although the probability of geopolitical conflict is relatively low in most protected areas, there is a significant percentage of the world's protected areas (>10%) located in areas where such risks are high.

Protected area management in areas at risk of geopolitical conflicts should develop management strategies to build capacity prior to conflicts while coordination, communication and collaborations are still possible (Debonnet and Hillman-Smith, 2004). Detailed in Table 3.7, the suggested management strategies emphasise increasing the resilience of economic, political and security functions which are threated during geopolitical conflicts.

Collaborating with local and international actors, diversification of funding sources, as well as outreach and education to ensure financial viability, self-governance and continued enforcement within protected areas are key actions that will determine the resilience of protected areas in times of turmoil (Debonnet and Hillman-Smith, 2004).

Areas at risk of geopolitical conflicts should also assess threats and conduct scenario mapping regularly and ensure that staff are trained to continue operation in times of conflict (Oglethorpe, 2004).

Finally, in light of the continued vulnerability of protected areas to threats post-conflict, protected area management should establish strategies to garner support to ensure they rapidly regain the necessary functions and funding for fully resumed operations (Debonnet and Hillman-Smith, 2004). This could be achieved by mobilising political support for the conservation of protected areas from all parties involved in the conflict, initiating new international alliances to re-establish strong transboundary cooperation.

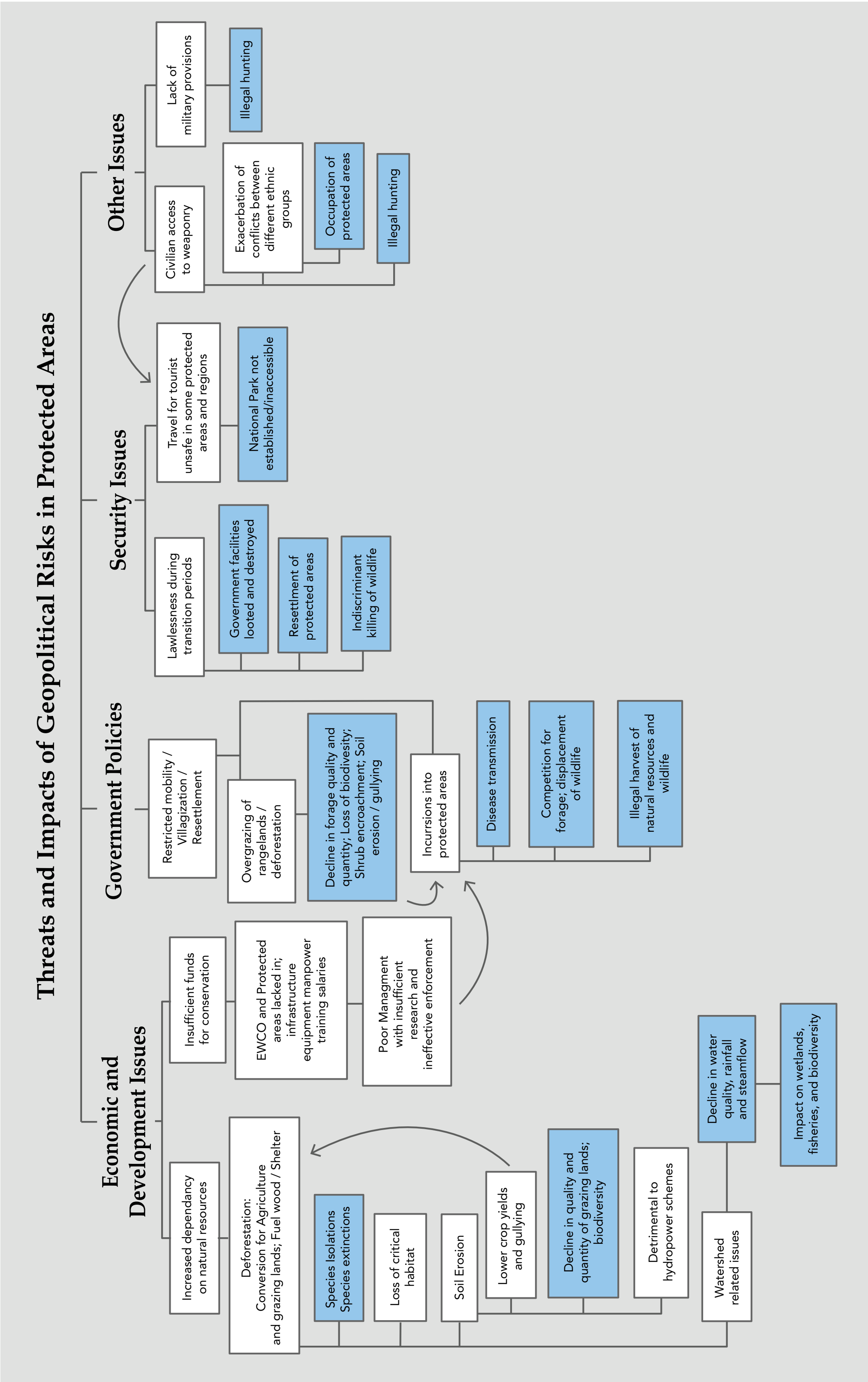


Figure 3.8: Threats resulting from geopolitical conflicts, the dysfunctions they cause and pathways leading to negative impacts in protected areas (modified from Jacobs and Schloeder, 2001)

Table 3.7: Summary of management strategies to address the risk of geopolitical conflicts

Functions threated by geopolitical conflict	Possible management strategies
Economic and Development Issues	Decentralization of conservation and protected-authority and assets Maintaining sound and diverse funding sources and financial systems. Collaboration with relief and development sectors.
Government Policies	Enable local communities to participate in the development of protected area management strategies and the establishment of locally-controlled community committee
Security Issues	Capacity building, scenario mapping and maintaining presence in protected areas during conflicts. Communicating with the military and line ministries.
Other Issues	Environmental education, outreach and creating awareness. Continued presence and enforcement in protected areas in times of conflict. Initiating new international alliances to re-establish strong transboundary cooperation

3.1.7 Illegal Wildlife Trade and Poaching

3.1.7.1 Introduction

The illegal wildlife trade refers to the illicit trading of wildlife specimens, in part or in whole (Rosen and Smith, 2010), and very often involves the trade of endangered species at risk of extinction (TRAFFIC, 2008; Zimmerman, 2003). The illegal wildlife trade is the second largest illegal trade in the world after narcotics and greater than illegal arms and ammunition (Zimmerman, 2003; Primack, 2010). It is one of the biggest threats to the survival of highly threatened species and is the second greatest threat after habitat destruction for many species (WWF, 2014).

The conditions of illegal trade are often unscrupulous, unsafe, and physically destructive to individual organisms and/or the surrounding environment (TRAFFIC, 2008). For example, illegal harvesters and poachers may use cyanide to kill fish, poor logging techniques, log within protected areas, use explosives in blast fishing on coral reefs, or use illegally obtained weapons and arms for poaching (TRAFFIC, 2008; Primack, 2010). Illegal trade undermines a nation's efforts to manage natural resources sustainably and can result in the loss of important earnings (TRAFFIC, 2008).

The illegal wildlife trade and poaching have been identified as serious risks to protected areas (Primack, 2010; IFAW, 2013; Anderson and Jooste, 2014; WWF, 2014). Not only do they pose a severe threat to conservation and biodiversity, they threaten the popularity and safety of tourism in PAs, decrease the value of their biophysical assets, and can unbalance their ecosystems.

Poaching has led to serious declines in already endangered species and deeply rooted iconic species, such as tigers (Panthera tigris), which once numbered 100,000 across Asia. Today, there are fewer than 3,200 tigers in the wild, with only 1,000 breeding females remaining, (WCS, 2013). Poaching has also led to the extinction of species and subspecies, and the extinction of species from specific countries. For example the rhinoceros (Diceros bicornis) completely disappeared from the wild in Mozambique in 2012 (Anderson and Jooste, 2014). Safaris and tourism are important foreign currency earners for African countries; Kenya, for example, earns over $1 billion annually from tourism (Anderson and Jooste, 2014). These revenues will be severely affected if visitors encounter fewer animals as a result of increased criminality in game parks and reserves (Anderson and Jooste, 2014).

Poaching is a major risk for protected areas. In May 2013, CITES and UNESCO expressed concern over increased poaching and specifically the killing of elephants (Loxodonta spp.) in Dzanga-Sanga National Park, part of Sangha Trinational, a World Heritage Site located at the borders of Cameroon, Congo, and the Central African Republic (UNESCO, 2013). Remote and relatively well-protected World Heritage Sites are targets for mass killings of elephants (IFAW, 2013). The Okapi Wildlife Reserve in the Democratic Republic of Congo is a World Heritage Site that has been targeted by elephant poachers in the last two years. In June 2012, mai rebels, a local gang made up of elephant poachers and illegal miners, attacked the reserve's headquarters and neighbouring village, killing six local people (including park rangers) and 14 rare okapis (Okapia johnstoni) (Okapi Conservation Project, 2012; IFAW, 2013).

The UN has now classified timber and animal trafficking as serious organised crimes (IFAW, 2013). The illegal wildlife trade attracts the attention of organised criminal syndicates and has been connected to a number of other black market industries that follow similar patterns (Zimmerman, 2003). Wildlife traffickers use smuggling routes similar to those of drug traffickers and the crimes often become entangled, with smugglers branching out into animal trafficking in order to mask their drug trafficking (IFAW, 2013). Many of the crime syndicates carry out detailed planning, have significant financial support, are often well armed, and understand and use new technology from military weapons to helicopters, (IFAW, 2013).

3.1.7.2 Geographical context

Trade in endangered wildlife has been a concern on the global environmental agenda since the dawn of international environmental law in the 1970s (Zimmerman, 2003; Beyerlin and Marauhn, 2011). Trade in wildlife is an issue central to the relationship between biodiversity and sustainable development. Increasing demand, consumption, and the desire for international commercial trade entail the depletion of many natural living resources at a faster rate than they are being replenished (Broad, Mulliken and Roe, 2003). For this reason, a number of international treaties have been established to mitigate and control the sale of wildlife internationally (Birnie, Boyle and Redgwell, 2009). CITES and IWC are international treaties that aim to prevent unsustainable wildlife trade by making the harvesting, poaching or trade in wildlife illegal, and have seen success. Box 3.2 and 3.3 explain these treaties.

Most major exporters of illegal wildlife are in the developing world, often in the tropics, especially Africa. The vast majority of the illegal poaching and export of wildlife takes place in developing countries and in emerging markets, such as BRIC countries, as wildlife capture is difficult to monitor with limited resources, and not always punished. The purchase and trade of illegal wildlife occurs both internally and transnationally within developing and developed countries (Zimmerman, 2003; IFAW 2013). East Asia in particular stands out as a purchasing hub of illegal wildlife trade. China, Hong Kong, Japan, Singapore, and Taiwan are major importers of illegal wildlife, but Canada, the European Union and the United States are also responsible for importing illegal wildlife. A major concern is that the vast majority of seized illegal material is indistinguishable from legally traded wildlife goods or farmed substances, yet these are often sold in the same market or contribute to the same commercially sold products (Zimmerman, 2003; IFAW, 2013).

Box 3.2: *CITES*

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) established in 1975 is one of the most successful international environmental treaties (Leader-Williams, 2003; Zimmerman, 2003). CITES regulates international trade in certain wildlife species, mainly those that are or may become threatened with extinction. CITES lists the species of animals and plants it strives to protect in three Appendices. Presently, 124 countries are parties to CITES Appendices listing protected species, including the United States, Hong Kong, China, Japan, and South Korea. The international trade of species listed in the Appendices is prohibited without a CITES permit (Lee, 1996). Each party to the Convention must establish at least one Management Authority and Scientific Authority responsible for ensuring that conditions are met before permits are issued (Lee, 1996). The effectiveness of the treaty is greatly undermined by the illegal wildlife trade.

Box 3.3: *IWC*

Commercial whaling is the single most significant threat to larger whale species, from which many have not yet recovered (Morrel, 2007). Hunting of right and gray whales was declared illegal in 1935 by international agreement. By this time their populations had been reduced to less than 5% of their original abundance. The International Whaling Commission (IWC) was established in 1946 by whaling nations to sustain whale hunting. In the early 1960s the IWC began to institute partial bans on whaling for parts of the world and certain species. In 1986 the IWC instituted a moratorium on all commercial killing of whales worldwide, in the face of the opposition of Japan, Norway, Russia and Iceland. These nations have continued hunting despite the IWC agreement (Primack, 2010).

3.1.7.3 Materiality

There are five living species of rhinoceros, and all are among the world's most endangered large mammals (Leader-Williams, 2003). Three species exist in Asia (Sumatran, Javan and Indian; Dicerorhinus sumatrensis, Rhinoceros sondaicus, and Rhinoceros unicornis respectively) and two species exist in Africa (white and black; Ceratotherium simum and Diceros bicornis respectively). All of these species are critically endangered, with some sub-species perilously close to extinction. The world populations of southern white and Indian rhinos are in the low thousands. The population of all rhinos has declined from hundreds of thousands in the early 1900s to 75,000 in the early 1970s to 13,000 (Leader-Williams, 2003). Although there are multiple reasons for the loss of rhinos (including habitat loss and legal hunting for sport), illegal hunting of both protected and unprotected populations due to the demand for rhino horn and its trade has been labelled largely responsible for their reduction (Leader-Williams, 1992; Leader-Williams, 2003).

The African elephant (Loxodonta africana), listed under CITES Appendix I (commercial trade not permitted), has undergone a long period of very high levels of illegal and unsustainable hunting. Ivory has been banned from commercial trade since 1990. Waves of illegal poaching in the 1970s onwards were due, among other factors, to the collapse of the purchasing power of currencies and wages in multiple African nations (Jachmann, 2003). Scarce economic opportunities led to the commercialisation of illegal ivory and rhino horn (Jachmann, 2003). In South Luangwa National Park, elephant numbers had dropped to about 15,000 in 1987. In 1988 the population declined further to 2,400, due to illegal poaching and the movement of elephants away from the affected area (Jachmann, 2003).

Elephants and rhinoceros provide a key source of revenue for ecotourism; many tourists are keen to see elephants when visiting PAs (Duffy et al, 2010), and they are an important flagship species for conservation (Walpole et al, 2002). A PA with large populations of endangered species, such as rhinoceros and elephants, could symbolise a healthy, and therefore successful, protected area, which is likely to encourage more tourism. The loss of elephants would directly correlate with a reduction in the number of tourists; high crime rates from poaching could also deter tourists.

Bush meat is wild meat hunted for human consumption. This includes large and small mammals and birds. Levels of bush meat trade in the tropics and in West and Central Africa have become unsustainable. Bush meat is now considered a threat to poorly monitored protected areas (Bowen-Jones, 2003). Habitat loss and fragmentation exacerbate the problem of unsustainable levels of bush meat hunting (Bowen-Jones, 2003).

During the 1970s the marine ornamental fish trade expanded into a commercial, multimillion dollar industry, with fisheries operating across the tropical world (Woods, 2001). About 45 countries supply the market, important suppliers being Indonesia, the Philippines, Brazil, the Maldives, Vietnam, Sri Lanka and Hawaii. The main consumer markets are the United States, Europe and East Asian countries, particularly Japan. The total import value of marine ornamental specimens is estimated to be US$28 to 44 million (Wood, 2001). In terms of the global annual catch, this value could range from 14 million to more than 30 million fish (Wood, 2001). As well as ornamental fish, much illegal fishing for the food trade also occurs worldwide (Agnew et al, 2009). The methods used to capture fish are often highly destructive to ecosystems, including cyanide poisoning, blast fishing, long lines, and illegal netting. Cyanide causes high post-harvesting mortalities, blast fishing can destroy three-dimensional coral ecosystems, while the other methods can cause high mortality of non-targeted and uncaught species, or bring in high levels of by-catch which are then discarded (Speight and Henderson, 2010). The legality, registration and licensing of fishing within various sectors of the industry often goes unchecked, leaving opportunities for illegal fishermen and collectors to unsustainably over-exploit populations, to capture species with low survival rates, and to employ destructive methods (Wood, 2001).

3.1.7.4 Manageability

As well as having serious impacts upon biodiversity, the illegal wildlife trade is a source of many institutional risks such as 'eroding state authority' and 'fuelling civil conflict' which threaten 'national security' and incur 'substantial economic losses internationally' (Lawson and Vines, 2014). Many impacts on political and national security remain indirect and unknown, and therefore continue to be poorly managed.

The network of actors in illegal wildlife trade involves harvesters and poachers, armed non-state actors, international crime groups and corrupt institutions and institutional actors across a global network, including large organised crime syndicates, individual non-state actors and 'legitimate' authorities and state-actors (Lawson and Vines, 2014; Oldfield, 2003; Goss and Cumming, 2013). Network analyses indicate that numbers of participants within local trade networks are continuing to increase rapidly (Goss et al). Former US Secretary of State Hillary Clinton, in 2012, described the illegal wildlife trade as a 'global challenge that spans continents and crosses oceans', and 'a national security issue, a public health issue, and an economic security issue'. Illegal wildlife trade is seen as both a conservation issue as well as a crime threatening the legitimacy and security of institutions and societies at various points in the networks of actors (Lawson and Vines, 2014).

Management of the illegal wildlife trade is an ongoing challenge, particularly due to its criminality and often violent nature. The difficulty in monitoring rainforest populations due to their inaccessibility and in catching poachers in the act limits the legal control that is possible over unauthorised hunting and poaching. Military strategies have frequently been used and have often been the most successful in targeting organised criminal groups or stopping them in their tracks (IFAW, 2013). Treaties and trade bans such as CITES help to decrease the rate of illegal trade between countries, and other non-military strategies for reducing illegal hunting, where its impact is relatively minor, have been increasing in popularity. When local villagers or indigenous communities are involved in hunting, such as for subsistence, military strategies are often not the most effective approach and also expose protected area management to controversy. Successful PA management of animal populations may therefore involve a combination of military and non-military strategies, depending upon the actors involved.

Many protected areas have already taken measures to reduce poaching and illegal harvesting. Such measures include penalties, imprisonment, fines, military action within parks, and confiscation at borders. However, much hunting still goes undetected and unpunished due to the difficulty of finding culprits, corruption, or nations simply choosing to ignore trade bans (Zimmerman, 2003; IFAW, 2013).

Military strategies have been advised for approaching large criminal syndicates. Militia or armed rangers should be included in management plans when rangers are at risk of being killed by poachers, when the poachers are highly armed with modern technology (from helicopters to AK-47s), or when the animal death toll is particularly high, such as hundreds or thousands of elephants slaughtered a year (IFAW, 2013).

Other strategies could include cooperation between leaders of the international community: both supply and demand countries. Penalties for serious crimes (such as large-scale shipments or organised criminal activity) could be increased, as the lenient approach to wildlife crimes has very often been insufficient to deter illegal trade activities (Oldfield, 2003).

Empirical data on the illegal wildlife trade, statistics of trade, and details about the motivations of the actors involved are limited (Lawson and Vines, 2014). Evidence on how the severity of punishment and the strictness of policies deter illegal wildlife trade is of importance for crafting new protected area policies. For example, in 1989, the African elephant (Loxodonta spp.) was moved from CITES Appendix II to Appendix I, which banned the international trade in ivory among member states. However the absence of legal trade in ivory coincided with a spike in the uncontrolled black market trade to meet demand through illegitimate and unmonitored means (Lemieux and Clarke, 2009). Such long-term implications need to be analysed in detail to sustain effective protected area management and international trade policies.

Educational policies to inform subsistence traders are important for creating a support system for low-income communities dependent on wildlife trade for their livelihoods. Evidence from Laurance (2014) also suggests that the mere presence of field researchers can help to limit illegal poaching and over-harvesting threats; and their presence can also provide economic benefits by generating support for protected areas through research.

3.1.8 Infrastructure Development

3.1.8.1 Introduction

Infrastructure is essential for the production and distribution of goods and services, and includes transport (roads and railways), energy production (power plants), raw materials (mining of minerals, extraction of oil and gas), communications (cell phone masts, power pylons) and tourism (visitor centres, viewing towers, crossings).

However, environmental concerns are seldom considered during the design, planning and construction of infrastructure projects. The mismanagement of infrastructure development has significant impacts on local environments (Laurance et al, 2001; Fearnside and Graça, 2006; Blake et al, 2007; Benítez-López, 2010; Laurance et al, 2010; Laurance et al, 2014; Haddad et al, 2015; Laurance et al, 2015). For instance, poor planning of a road upgrade project in the Dja Faunal Reserve in Cameroon increased the instances of logging and poaching (Rice and Counsell, 1998). Finer et al (2008) also reported on how oil and gas development in the protected areas of western Amazon has resulted in deforestation for access roads, drilling platforms and pipelines, and contamination from oil spills and wastewater discharges. Large dams have also affected migratory fish, spawning habitats, aquatic biodiversity, fisheries and riverine communities, due to the considerable disruptions they cause to the hydrological and biological characteristics of free-flowing rivers (WWF, 2006).

In addition, unequal distribution of costs and benefits from infrastructure projects have also been recorded, resulting in negative social impacts of unsuitable infrastructure development on the indigenous and marginalised people, as well as rural and forest-dwelling communities through increased exposure to new social and market pressures, relocation and the loss of land (Raman, 2011). Fearnside (2002) described how improved roads constructed in the Great Nicobar Islands East made chewing tobacco and alcohol accessible to the indigenous Shompen population. Excessive consumption of tobacco and alcohol has subsequently led to considerable health and social issues in the community and undermined their previously self-sufficient lives (Fearnside, 2002).

At the annual G20 summit in 2014, nations committed to invest US$60-70 trillion worldwide in new infrastructure by 2030, which would more than double current global infrastructure (Alexandre, 2014). Hence, infrastructure development is an impending threat that necessitates preventive management action in order to safeguard protected area values.

3.1.8.2 Materiality

Infrastructure developments that affect protected areas can be classified under four main headings: i) public works; ii) oil and gas; iii) mining; and iv) tourism (WWF, 2015). The general threat these infrastructure developments pose to protected areas include downgrading, downsizing and degazettement, degradation and destruction of natural habitats, increased accessibility and transformation of landscapes (Table 3.8).

However, due to the differences in physical structures and activities involved the potential negative impacts they have on protected areas vary. For example, public works tend to involve the creation of transport links, connecting protected areas to urban areas. Studies have shown that such connections to urban centres present prime habitats for invasive species and contribute significantly to their establishment in protected areas (Jolly et al, 2011; Meunier and Lavoie, 2012). The clearing of land for public works also leads to habitat loss and fragmentation and the transformation of landscapes. The resultant edge effects increase temperature and decrease humidity, leading to desiccation of vegetation. Similarly, oil and gas extraction and mining infrastructures require land clearance and access roads causing habitat loss and fragmentation (see Section 4.1.3 for more on threats from extractive industries). More importantly, these activities also generate considerable waste, often leading to severe pollution issues (Edwards et al, 2013). Finally, tourism in addition to increasing habitat loss and fragmentation, also increases the flow of people and associated activities into protected areas and can significantly change the behaviour of animals (Constantine et al, 2004; Johnston, 2006).

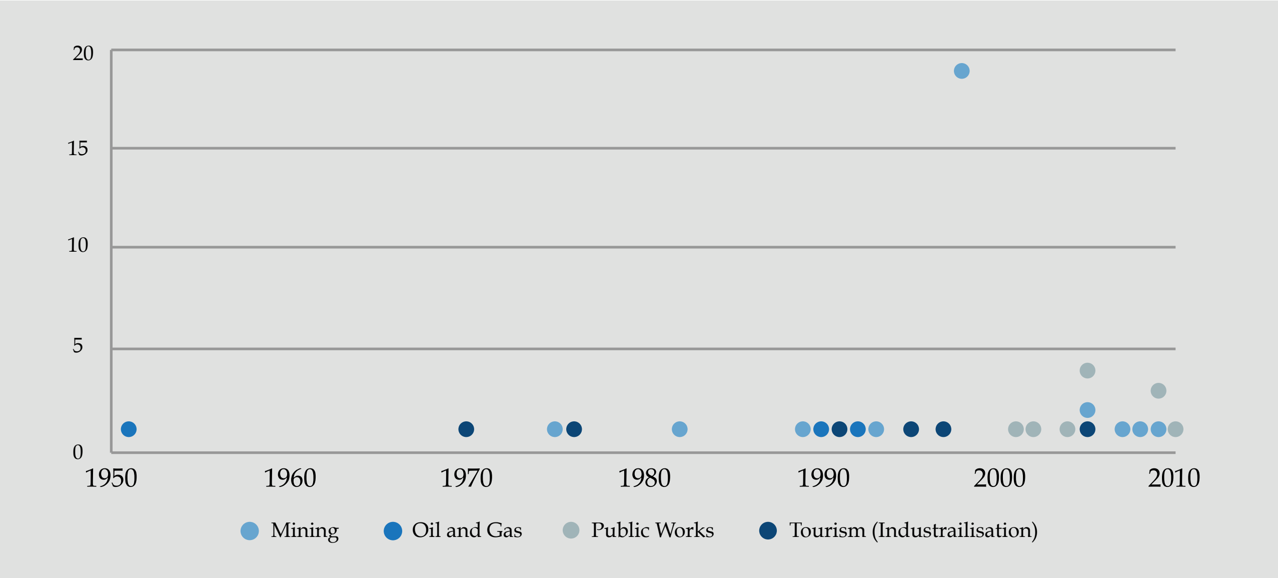
***Table 3.8:** Types of infrastructure development, corresponding threats and the potential negative post development impacts*

Type of infastructure development	Components	Threat	Potential Negative Impacts
Public Works	Dams, roads, railways, pipes, electrical grid, power-generation facilities, telecommunications towers, transportation facilities	Downgrading, Downsizing, and Degazettement	Spread of invasive alien species
		Degradation and destruction of habitats	Desiccation of vegetation
		Increased accessibility	Habitat loss and fragmentation
		Transformation of landscape	Habitat loss and fragmentation
Oil and Gas	Fields, wells, storage tanks, refineries, pipelines	Downgrading, Downsizing, and Degazettement	Severe habitat loss and fragmentation
			Increased pollution
Mining	Open-pit mines, underground mines, riverbed mines, quarrying, subsurface mines, and related physical structures for the extraction of metals, minerals, coal, rock, stone, sand,	Downgrading, Downsizing, and Degazettement	Severe habitat loss and fragmentation
			Increased pollution
Tourism	Visitor Centres, roads, accommodation	Degradation and destruction of habitats	Spread of invasive alien species Changes in animal behaviour
		Increased accessibility	Increased garbage, pollution
		Transformation of landscape	Habitat loss and fragmentation

Despite the growing literature highlighting the negative impacts of infrastructure development and the need to carefully consider its necessity in or near protected areas, demand for public works infrastructure and natural resources continues to rise. According to monitoring data of PADDD events, obtained from the WWF PADDD Tracker (2015), instances of these events related to infrastructure development increased considerably from 1950 to 2010 (Figure 3.9), shifting from oil and gas and mining infrastructure development in the past, to tourism and public works related infrastructure development in recent years (Figure 3.10).

In addition, the PADDD data revealed that these infrastructure developments are not restricted to unclassified protected areas or those that are designated for the sustainable use of natural resources, tourism and recreation, education and maintenance of cultural and traditional attributes (IUCN Category V and VI), but also extend to strict protected areas preserved for the sole purpose of scientific research and wilderness protection (IUCN Category Ia, Ib and II). This worrying trend, which signals the weakening of conservation commitments in the face of the pressure of development, is exemplified in recent proposals from New Zealand. Haggart (2009) revealed that the New Zealand government has plans to downgrade an unspecified number of protected areas, including Mt Aspiring National Park despite its status as a World Heritage Site

Figure 3.9: *PADDD caused by mining, oil and gas, public works and tourism from 1950 to 2010.*



Source: WWF. 2015. PADDDtracker.org Data Release Version 1.0 (January 2014). Washington, DC: World Wildlife Fund.

Wittemyer et al's (2008) study on the borders of 306 PAs in 45 countries in Africa and Latin America suggested that rapidly increasing human populations around protected areas increase demand for public works infrastructure and utilisation of natural resources. As world population approaches the projected nine billion by 2050, demand for land for agriculture to feed them is expected to grow and will threaten protected areas (Green et al, 2005). In addition, the bulk of the population growth will occur in developing countries where current infrastructure is unable to provide for a rapidly expanding population. McNeely and Schutyser (2003) pointed out that this would promote further urbanisation projects, which will begin to encroach into protected areas. Moreover, rising incomes in these developing countries are expected to increase consumption (Price et al, 2006) and lead to the demand for more recreational spaces in protected areas, exposing them to the threat of infrastructure development (McNeely and Schutyser, 2003).

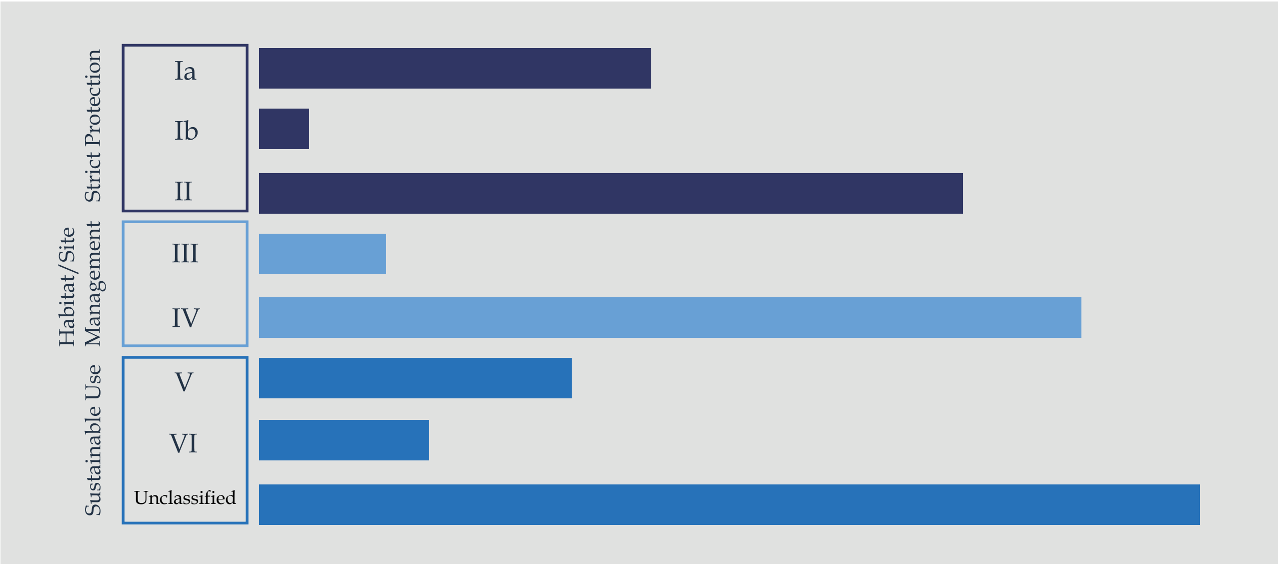
3.1.8.3 Manageability

Considering the expected continued increase in infrastructure developments and the range of negative impacts they cause to PAs, it is crucial for PA management to take a more proactive role in the assessment of projects and to adopt more stringent criteria that consider both short-term and long-term effects.

A PA management team should engage investors and key stakeholders during early proposal stages, to review the design of the projects and evaluate the necessary changes or protest against the developments if they are deemed to have long-lasting negative impacts. This is especially crucial for projects involving the creation of extensive road networks that have huge secondary and tertiary impacts. Recent 'offshore' natural resource projects that operate without associated road networks have been conducted in the Peruvian Amazon, with all personnel being transported to and from the site by helicopter. Pipelines that carry natural gas were buried and the area above them restored.

In addition, PA management should prepare maps of social and biodiversity indicators and natural values, accurate spatial data on roads and other infrastructure within the protected areas and make these data publicly available to facilitate planning and evaluation of infrastructure projects. Such data will allow decision-makers to have a clear idea of areas of high value and assist them in determining suitable, compatible infrastructure developments. Such data can help to advance integrated land-use planning, which is a near-term priority, considering the anticipated growth in population.

Figure 3.10: *PADDD cases as a result of infrastructure, mining and oil and gas*



Source: WDPA. 2014.

Finally, considering the great pressures governments and project proponents with vested financial interests place on major development banks to approve infrastructure proposals, input from non-governmental environmental and social welfare groups and the general public, especially those directly affected by the project, is crucial in order to balance these pressures. PA management should hold frequent focus group discussions with NGOs and the public to communicate any proposed infrastructure developments and take their views into consideration. Also, PA managers could collate a list of socially necessary or acceptable infrastructure projects agreed upon by local and national stakeholders.

3.1.8.4 Manageability: What PA management can do to manage these risks

Decisions on where infrastructures are constructed are almost always politically driven and will be challenging for protected area management to change once the regulations have been passed. Hence, in order to ensure high-level political commitment, there is a need to promote public education, communication and awareness. In addition, constant review of values and benefits and the potential threat of development plans around and within protected areas ensure preparedness to react to future proposals.

Managers should also consider the risk of infrastructure developments in conservation planning within protected areas. For instance, restoration and rehabilitation efforts should be focused in areas which are less threatened by infrastructure developments and shifting of core conservation areas out of areas which coincide with high quantities of natural resources should also be considered, especially in parks designated for sustainable use.

Protected areas managers should also build up their capacity to perform environmental and social impact assessments and be familiar with tools such as 'mitigation hierarchy' and biodiversity offsets so they can rapidly respond and advise on planned infrastructure developments, and suggest just compensation when these developments are inevitable.

Finally, protected areas managers should also work closely with local communities and government to effect legislation that safeguards the values of protected areas from those infrastructure developments that are incompatible with the values of the management objectives. This is particularly crucial for the many protected areas that do not have clear management goals and legal status.

3.1.9 Jurisdictional Tensions

3.1.9.1 Introduction

Jurisdictional tensions can be understood as strained political relationships where parties possess differing authority to decide on legal cases, resulting in disharmony. They typically arise due to power relations. This is dependent on the specific institutional arrangement that determines assignment of powers and responsibilities in respective countries (Lausche, 2011).

Although jurisdictional disputes are a phenomenon intrinsic to bureaucracy, they tend to be particularly prevalent in protected areas. This is because administrative responsibility in PAs seldom falls within the realm of one ministry or department. In addition, the designation of authorities tends to be unclear where powers and responsibilities of multiple ministries have differing ideals and aims for resource use in the same area.

Disputes over jurisdiction in protected areas divert park management resources into funding battles, break down attempts to integrate sector policies and damage economic links. Unaddressed, the ensuing uncertainty could lead to the loss of confidence of donors and investors and the polarisation of public opinion, resulting in conflicts with authorities (see Section 3.1.3) and, in extreme cases, social unrest (see Section 3.1.11). Jurisdictional tensions have often been regarded as a peripheral concern as they do not directly affect protected areas (Lausche, 2011).

However, they have potential to indirectly affect a wide range of protected area values and so the ability to initiate low probability but high impact events. Hence, the situation calls for better understanding of the threats that jurisdictional tensions pose and the formulation of strategies to increase the resilience of protected areas against them.

3.1.9.2 Materiality

Although protected areas often fall under the charge of the ministry of environment, in many jurisdictions forests designated as reserves within the protected areas may be separately assigned to the ministry of agriculture, while the ministry for tourism may oversee areas set aside for ecotourism (Lausche, 2011). For example, in Thailand, the Ministry of Natural Resources and Environment has jurisdiction over managing protected forests, but the Ministry of Agriculture and Cooperatives has jurisdiction over logging and forest resource exploitation resulting in inter-agency conflicts over illegal logging and other forest issues (Hawkins, 2010).

Governance structure determines the decision-making and management authority of protected areas and provides a broad overview of their vulnerabilities to jurisdictional tensions. The risk of jurisdictional tensions increases with the number of stakeholders involved, ambiguity over responsibilities, absence of mediators and differences in interests within the governance structure. Looking at the four main PA governance types and their sub-categories as recognised by IUCN (Borrini-Feyerabend et al, 2013), Type A and B governance usually involve multiple stakeholders, occasionally with competing interests and are more vulnerable to jurisdictional tensions between ministries than Type C and D, which have clearer property rights and legislation to safeguard them (Table 3.9).

In addition to governance structure, the specific management objective and status of protected areas also determines their vulnerability to jurisdictional tensions. Status and management objectives define the degree of legislation and enforcement required and hence the actors involved. Strictly protected areas (IUCN Category Ia, Ib and II) are usually well covered by clear national laws that restrict human activities and infrastructure development, typically involving few actors (Mázsa et al, 2013). Meanwhile, the PA management team usually takes the helm in areas that are conserved for environmental services, specific cultural and natural features and the preservation of species and genetic diversity (IUCN Category II, IV). Clear legislation and enforcement are typically lacking in unclassified protected areas or those that are designated for the sustainable use of natural resources, tourism and recreation, education and maintenance of cultural and traditional attributes (IUCN Category V and VI) (Jachmann, 2008). In addition, it is conventional to have multiple stakeholders and ministries involved in these protected areas (see Table 3.9), making them considerably more susceptible to jurisdictional tensions.

Rapidly increasing world population sparks fierce competition among stakeholders not only for natural resources, but also for scarce political resources (Xu and Melick, 2007). Cases demonstrating these emerging multiple use issues over forestry, tourism, agriculture, hunting, extraction, fisheries, transport and energy were clearly detailed in Bouwma et al (2010) report on Natura 2000 sites across Europe. Recent studies in the protected areas of Peru (Scullion et al, 2014) and Guatemala (Blackman, 2015) similarly found increasing tensions in protected areas proximal to localities experiencing increases in population and development. These studies indicate a possible link between demographics and jurisdictional tensions and highlight the importance for protected area management to monitor population trends.

Finally, in the last 20 years, protected areas have seen a movement towards plural management and governance models and growing emphasis on larger PAs, transboundary PAs, connectivity conservation and landscape approaches (Dudley et al, 2014). As previously discussed, both governance type and management category of parks affect which jurisdictional tensions are likely to occur. Hence, the shift towards diverse management and governance models may introduce novel cross-jurisdictional policy issues in existing protected areas and expose new areas to the risk of tensions.

Table 3.9: *IUCN governance types and subcategories*

Governance type	Sub-categories
Governance by government	Federal or national ministry or agency in charge
	Sub-national ministry or agency in charge
	Government-delegated management (e.g. to an NGO)
Shared governance	Transboundary management
	Collaborative management (various forms of pluralist influence)
	Joint management
Private governance	Declared and run by individual landowners
	By non-profit organisations e.g. NGOs, universities)
	By for profit organisations (e.g. Corporate owners, cooperatives)
Governance by indigenous peoples and local communities	Indigenous peoples' protected areas and territories – established and run by indigenous people
	Community conserved areas – declared and run by local communities

Source: Borrini-Feyerabend et al (2013)

3.1.9.3 Manageability

Protected area management will have limited capacity to manage and facilitate discussions after the onset of jurisdictional tensions. Hence, it is necessary to set up an inclusive and democratic decision-making framework and process that serves the common interest in protected areas before issues surface.

The best way to identify common interest and ensure its security is to provide access to knowledge, and fair opportunity for voices to be heard in decision-making through improved participation (Lasswell, 1971; Brunner et al, 2002). Protected area management should improve opportunities for greater meaningful community and public involvement in genuine problem-oriented ways, reaching out not only to the participants identified but also to transient groups such as visitors. In addition to bridging local, public, and professional environmental knowledge and problem-solving for mutual advantage, such projects also ensure that perspectives and values are made transparent to authorities. This will facilitate more respectful and cooperative interactions between various ministries with jurisdiction in the protected areas and aid those entrenched

Also, appraisals of current decision-making processes by ministries and the degree of involvement and authority protected area management has in these decisions should be reviewed and improved (Brunner et al, 2002, 2005). There should be a shift away from technical, expert-driven, and conventional agency means of problem-solving, toward more integrative, inclusive problem-solving approaches. Such cooperative approaches ensure better identification and subsequent rectification of underlying procedural dynamics and conditions so that fair, effective and efficient decisions can be made within PAs.

Finally, there should be increased leadership coming from protected area management when addressing issues pertaining to resource preservation and ecological integrity within protected areas. Protected area management staff should strive to suggest changes to policies that fail to reflect the guiding principles and directions for resource management within the protected area systems. In addition, protected area staffs should be cognisant of the jurisdictional limits and tensions between the various ministries with authority in the park and constantly ensure that the projects and policies implemented by these authorities do not threaten the values of protected areas and adhere to the appropriate development and conservation practices within their jurisdiction.

3.1.10 Regulatory Changes

3.1.10.1 Introduction

Regulatory changes that affect protected areas typically encompass alterations in land use and land designation, often necessitated by the expansion of agriculture, mining and logging, infrastructure projects, land speculation, and urban residential or tourism development (Primack, 2010). These regulatory changes often involve legal authorisation to increase the number, magnitude, or extent of human activities (downgrading), excision of land or sea area through legal boundary changes (downsizing) and, in extreme cases, the complete retraction of legal protection, or degazettement, for entire PAs (Mascia and Pailler, 2011). Conservationists have been aware of these events since the beginning of the international conservation movement (Hance, 2011). However, because conservation research typically focuses on the regional or local levels, the extent of such regulatory changes leading to PA downgrading, downsizing and degazettement (PADDD) has only been recently highlighted by the global study conducted by Mascia and Pailler (2011).

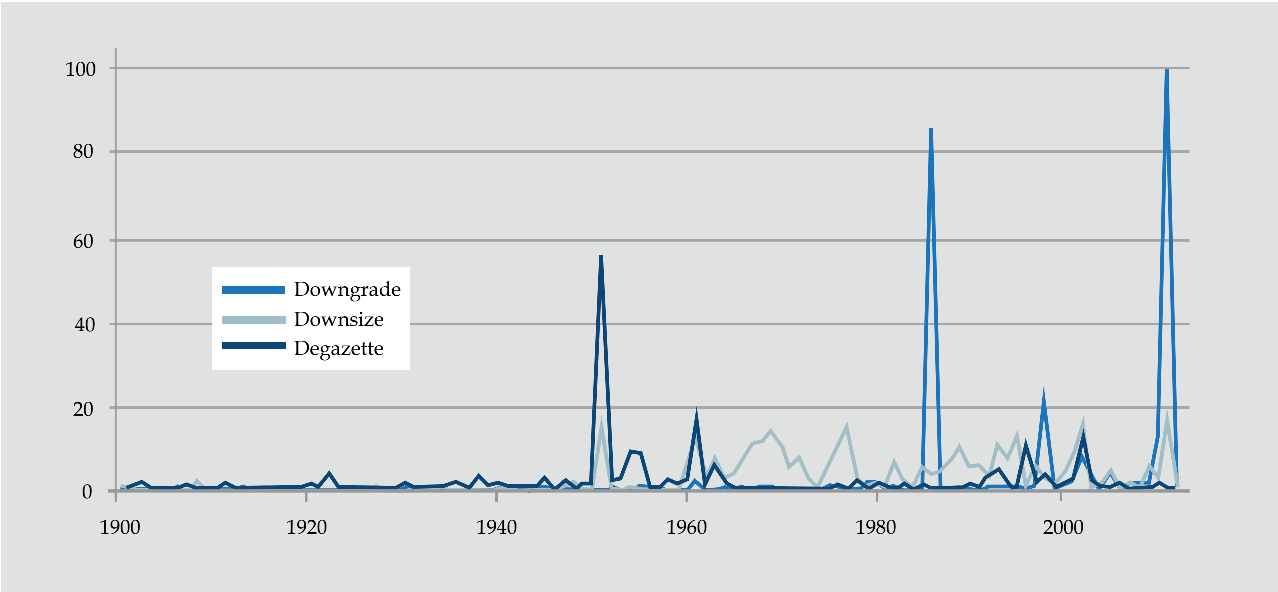
3.1.10.2 Historical context and geographical distribution

Since the 1900s, more than 2,000 events involving regulatory changes have been recorded (Mascia et al, 2014). These events affected an estimated 2,500,000km2 of protected lands and waters globally (Mascia et al, 2014). The number of recorded cases of regulatory changes were relatively low leading up to the 1950s, after which there was an exponential increase (Figure 3.11). Until the 1960s, these changes were mainly due to degazettement, after which the number of cases of downsizing increased considerably (Figure 3.11). Downgrading events were relatively uncommon and spiked in the early 1990s and late 2010s (Mascia and Pailler, 2011).

Contemporary studies have suggested that regulatory changes leading to PADDD can threaten the ecological integrity of PAs and potentially negate all previous conservation efforts within these areas (van Steenis et al, 1989; Walpole, 2003). For instance, the changes to regulations in peninsular Malaysia have been shown to be a key driver for forest loss, with rates of deforestation in PADDD areas faster than in forests that were not protected from 2000 to 2010 (Forrest et al, 2015). The estimated 1.5 million Mg of carbon lost from 2000 to 2010 translates to a worrying loss of US$4.5 to US$69.6 million in carbon stocks (Forrest et al, 2015). Likewise, in Cambodia, regulatory changes driven by the demand for industrial agriculture led to PADDD events that coincided with the sudden spike in deforestation rates observed in 2010 (Kroner, 2015). In addition to ecological impacts, Adams (2004) also highlighted the potential social impacts following regulatory changes due to the restrictions on access for certain groups of landowners and resource users.

The online PADDD-tracker launched in 2011 by WWF provides a database documenting the patterns, trends and causes of PADDD. In addition to keeping extensive records on enacted PADDD events, the database also allows users to update it with proposed PADDD cases. This information acts as a useful resource for identifying areas at risk of regulatory changes (see Figure 3.12).

Figure 3.11: *Temporal trends of global PADDD*



Source: WWF (2015)

We have noted that not all regulatory changes are bad and there have been cases where the downgrading of protected areas and surrendering of management rights to local communities have improved protection. For instance, Agrawal's (2012) research in India suggests that transitioning from strict protected areas to community-based systems that allow subsistence use of natural resources fosters local resource stewardship and enhances conservation outcomes. However, the number of these positive cases reported is relatively low (Mascia et al, 2014) and hence was not considered in our analysis.

More than 70% of all proposed PADDD events are located in the United States, Peru, Ecuador, Uganda, the United Kingdom, New Zealand and Australia, which collectively harbour more than 20% of the world's protected areas. There appears to be a considerable geographical shift in the risk of regulatory changes from developing countries to developed countries.

This rise in regulatory changes in developed countries was predominantly driven by budget cuts and rising demand for housing and recreation near nature areas (Watson et al, 2014). One such development is the a proposed high-speed rail link in the United Kingdom, the construction of which will potentially damage or destroy ten Sites of Special Scientific Interest – government-recognised protected areas, and nine NGO-run Wildlife Trust reserves (see Box 3.4 below). Ritchie et al's (2013) study of Australian protected areas demonstrated the rising trend of downgrading protected areas to allow for livestock grazing, mining and recreational hunting and fishing.

Meanwhile, the continued increase in regulatory changes in developing countries is attributed to the rush to exploit mineral wealth. This has resulted in at least five African nations downsizing or degazetting their national parks (Edwards et al, 2014). In Zambia, 19 national parks were degazetted to promote limestone mining, land in the heart of the Lower Zambezi National Park was even allocated to a huge copper mine (Edwards et al, 2014). Tanzania has also downsized Selous Game Reserve for uranium mining, while Guinea has downsized its Mt Nimba World Heritage Site for iron-ore prospecting (Edwards et al, 2014).

Box 3.4: *Case Study: Proposed high-speed rail link construction threatens PAs in the United Kingdom*

The proposed new high speed rail link from London to Manchester and Leeds via Birmingham has been designed to avoid densely populated areas and areas with built infrastructure as much as possible. The attempt to reduce the social impact of construction has led to ecological trade-offs. Construction plans of Phase 1 show at least nine nature reserves, ten Sites of Special Scientific Interest (SSSIs), and 153 Local Wildlife Sites (LWS) to be directly affected. In addition, a further 12 nature reserves, 17 SSSIs and 247 LWS will be potentially impacted indirectly by the construction. Regulatory changes allowing for the mitigation of impacts via biodiversity offsets have helped to justify for the intrusion into and downsizing of existing protected areas (Watkins et al, 2014).

Source: Watkins et al. (2014)

3.1.10.3 Materiality

Protected areas surrounded by intense land use, are more susceptible to regulatory change than those situated in remote and uninhabited regions (Hansen and DeFries, 2007). The number and types of intense land use further increase the likelihood of changes in regulations as competing demands drive authorities to amend regulations (Hansen and DeFries, 2007).

In addition, Hansen and DeFries (2007) also argued that the socioeconomic fabric of surrounding human communities contributes to the probability of protected areas and their surroundings being subjected to regulatory changes. Communities that benefit directly from the provisional goods and services of protected areas are more likely to resist regulatory change for cultural and survival reasons (Rasker and Hansen 2000). On the contrary, protected areas proximal to urban and suburban areas will be exposed to more frequent regulatory changes as the surrounding population is less connected to the land and hence puts up little resistance to these changes (Rasker and Hansen, 2000).

3.1.10.4 Manageability

As Mascia et al (2014) emphasised, the implications of regulatory changes on protected areas has only been recently recognised. Hence, the first step to manage the threat is to begin a vulnerability assessment within and around individual protected areas, based on the intensity and type of land use, socioeconomic fabric of surrounding human communities and the natural capital that the resources reflect. The assessment should aim to elucidate recent trends and gaps legislations and regulations that require reviewing.

In addition there is also a need for protected area management to work closely with local communities and government to understand and monitor shifts in values and perceptions before they are translated and formalised in regulations. Participation in these discussion will also ensure the compatibility of decisions and regulations with planned developments in the protected areas.

Finally, protected area managers should not view regulatory changes negatively and should be aware of the positive impacts they could provide. Regulatory changes may increase the protection status within parks, increase the amount of sites protected around parks and reduce the instances of incompatible developments and usage of the areas around protected areas. In addition, changes in regulations might also transform socioeconomic usage of the area allowing for improved integration of protected areas into the surrounding matrix. Protected area managers have to keep a look out for these opportunities as they present themselves to generate greater resilience in the areas they are managing.

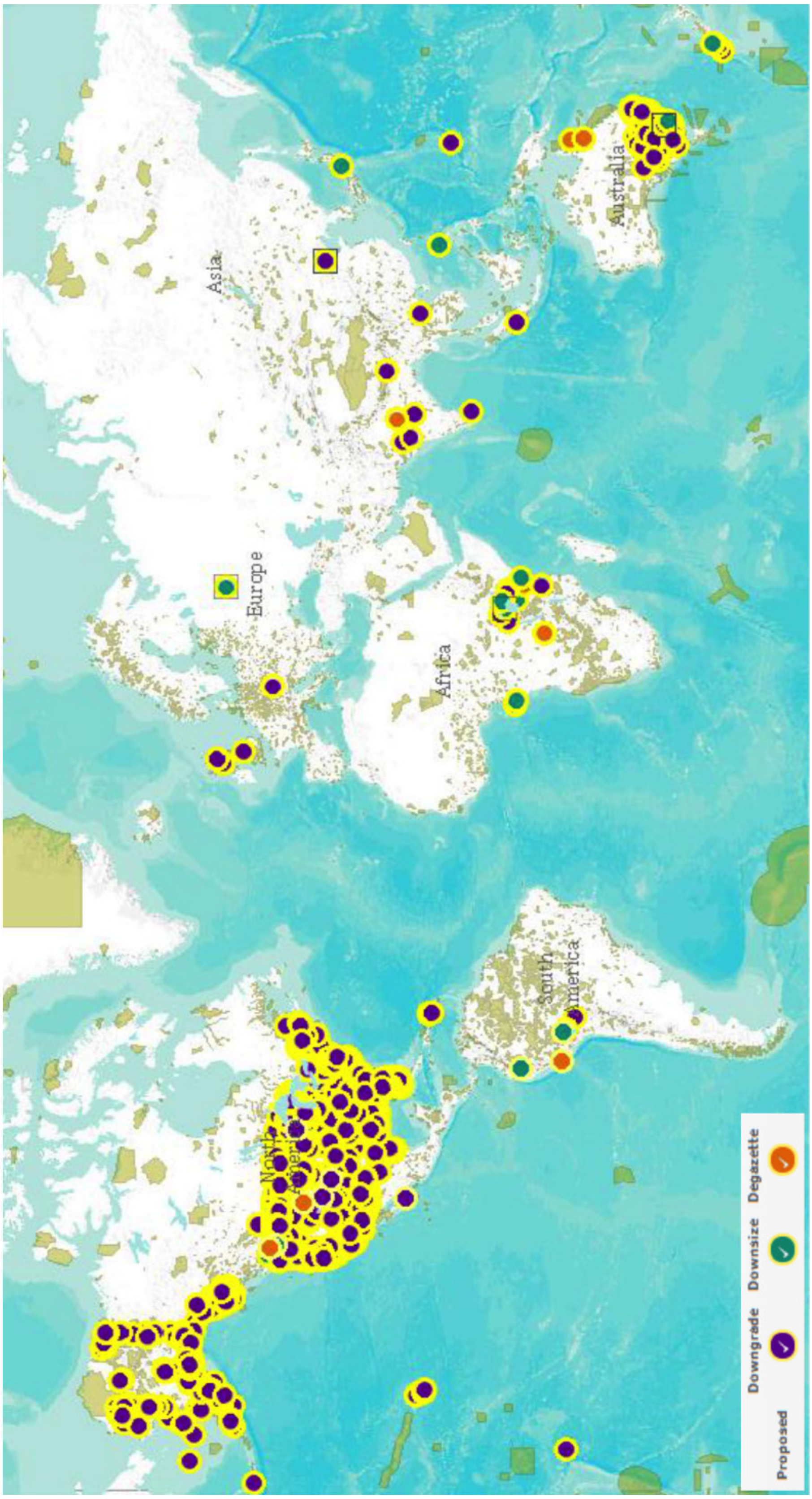


***Figure 3.12:** Global distribution of proposed PADDD indicating areas at risk of regulatory changes*

Source : WWF (2015)

3.1.11 Social unrest

3.1.11.1 Introduction

Social unrest is commonly expressed through protests, civil disorder, ochlocracy or insurrection. The world has experienced increasing social unrest in the past decade, which is believed to be driven by 1) structural causes such as rising deprivation, increasing high youth unemployment; 2) proximate causes associated with austerity measures by a government, negative food price shocks, racial and ethnic discrimination and oppression of civil rights; and 3) trigger causes like police brutality, environmental catastrophes and political and public events (Cambridge Centre for Risk Studies). Ortiz et al's (2013) study on international and national media identified a steady increase of protests, particularly after 2010. An analysis of data from 2005-14 extracted from the Global Database of Events, Language, and Tone (GDELT), concurred with the study by Ortiz et al, illustrating increased incidence of social unrest in the past decade.

A review of scientific journal articles revealed fewer than a dozen instances where social unrest and their respective impacts on protected areas were reported (Table 3.10). However, all the studies agree upon the considerable negative impacts, which range from the loss of habitats to assaults on protected area staff.

Further investigation into the threat social conflicts pose to protected areas and their values requires an understanding of their interaction across time. Studies agree that this occurs through the disruption of economic and development functions, government policies and the incitation of security issues. Table 3.10 lists how different types of social unrest can affect PAs. For instance, Eagles (2004) demonstrated the impact protests have had on tourism in national parks, while Draulans and Van Krunkelsven (2002) illustrated how insurgencies like the 1995-2006 civil war in the Democratic Republic of Congo (DRC) resulted in a significant loss of wildlife, including elephants, due to institutional collapse, lawlessness and unbridled exploitation of natural resources such as minerals, wood, ivory and bush meat. Table 3.11 summarises the degree of social unrest and the corresponding impacts collated from a series of studies by Draulans and Van Krunkelsven (2002); Debonnet et al (2004); Eagles (2004); Baral and Heinen (2005); and Helga (2013), detailing the various functions affected, the resultant dysfunctions and negative impacts the respective severity of social unrest has on protected areas. Considering the disruptive and destructive capacity that local conflicts possess and their recent proliferation, it is critical to assess the risks they present to protected areas values.

3.1.11.2 Geographical Distribution

The 2014 Social Unrest Risk Index (SU) produced by the Economist Intelligence Unit (2014) is a global attempt to represent the relative potential of local conflicts occurring in countries. Geographical distribution of protected areas in localities with high risk of local conflict is obtained by correlating data on protected area size with distribution data obtained from the UNEP-WCMC World Database for Protected Areas, with the SU.

The analysis revealed that 6.5% of the world's protected areas are located in the 22 countries with the highest risk of local conflict (SU Score = 4 to 5) (Figure 3.13). High risk of local conflict particularly coincided with large protected areas in the Latin American countries of Venezuela, Argentina, Bolivia and Brazil, and the sub-Saharan Africa countries of Congo, Guinea, Nigeria, Sudan and Zimbabwe, which have a history of political instability. However, due to their exposure to local conflicts, PA managements in these areas have received support from international actors and built up resilience through conservation programmes. For instance, the Maiko Tayna Kahuzi-Biega (MTKB) Landscape in Congo, comprising the Maiko and Kahuzi-Biega National Park has received the assistance of a consortium of NGOs and aid agencies including the USAID-funded Central Africa Regional Programme for the Environment (CARPE), Conservation International (CI), WWF, the Wildlife Conservation Society (WCS), the Jane Goodall Institute (JGI), and the Union of Associations for Gorilla Conservation and Development in Eastern DRC (UGADEC).

Approximately one quarter of the protected areas fall in areas of high risk of local conflict (SU Score = 3 to 4) (Figure 3.13). Of these, one third are located in Europe and Central Asia. Protected area management in these countries which include Spain, Portugal and Cyprus lacks the exposure to impacts associated with local conflicts, as well as funding and technical support, making them more vulnerable to the threats.

3.1.11.3 Materiality

There is often a progressive escalation from protest to civil disorder before unrest develops into mob rule and eventually insurgencies. Likewise, negative impacts increase progressively with intensifying conflicts, typically affecting tourism and access to funds at initial stages before accelerating to direct biophysical damage in later stages (Table 3.11). Multiple instances of such events have been recorded (Dudley et al, 2002; Debonnet & Hillman-Smith, 2004; Eagles 2004; Beall et al, 2013; Bown et al, 2013). In Honduras from 2008 to 2010, for example, economic crisis and an earthquake provoked social conflict which led eventually to deforestation and exploitation of marine resources (McSweeney et al, 2014, also see Box 3.4).

Box 3.4: *Escalating social unrest and impacts on the protected areas in Honduras.*

Protests following the 2008 global recession rapidly escalated into civil disorder after an earthquake of magnitude 7.1 hit in May 2009 (Brown et al, 2013). Subsequently, travel alerts and advisories were issued by countries including the United States, the United Kingdom, Canada and Australia (Haugen, 2009), leading to a decrease in tourists and tourism revenue of 9% in one year alone (Brown et al, 2013). The military coup which ensued in June 2009 further raised international concern about the stability of the government in Honduras. The World Bank swiftly suspended $135 million of loans for development programmes and various non-humanitarian foreign aid soon after. These funds included those which were used to develop and maintain the protected area programmes (Brown et al, 2014). Unable to support their functions and staff, the gradual breakdown of protected area management saw the influx of illegal loggers, cattle ranchers and land speculators encroaching into the protected areas and exploiting their resources (Malkin, 2014). Copén and a few other villages on the edge of the 1.3 million-acre Río Plátano biosphere reserve, once lauded as models for forest conservation, were spurned and the PA's land assets used to launder drug money. Habitat loss and local extirpation of species continue (McSweeney et al, 2014).

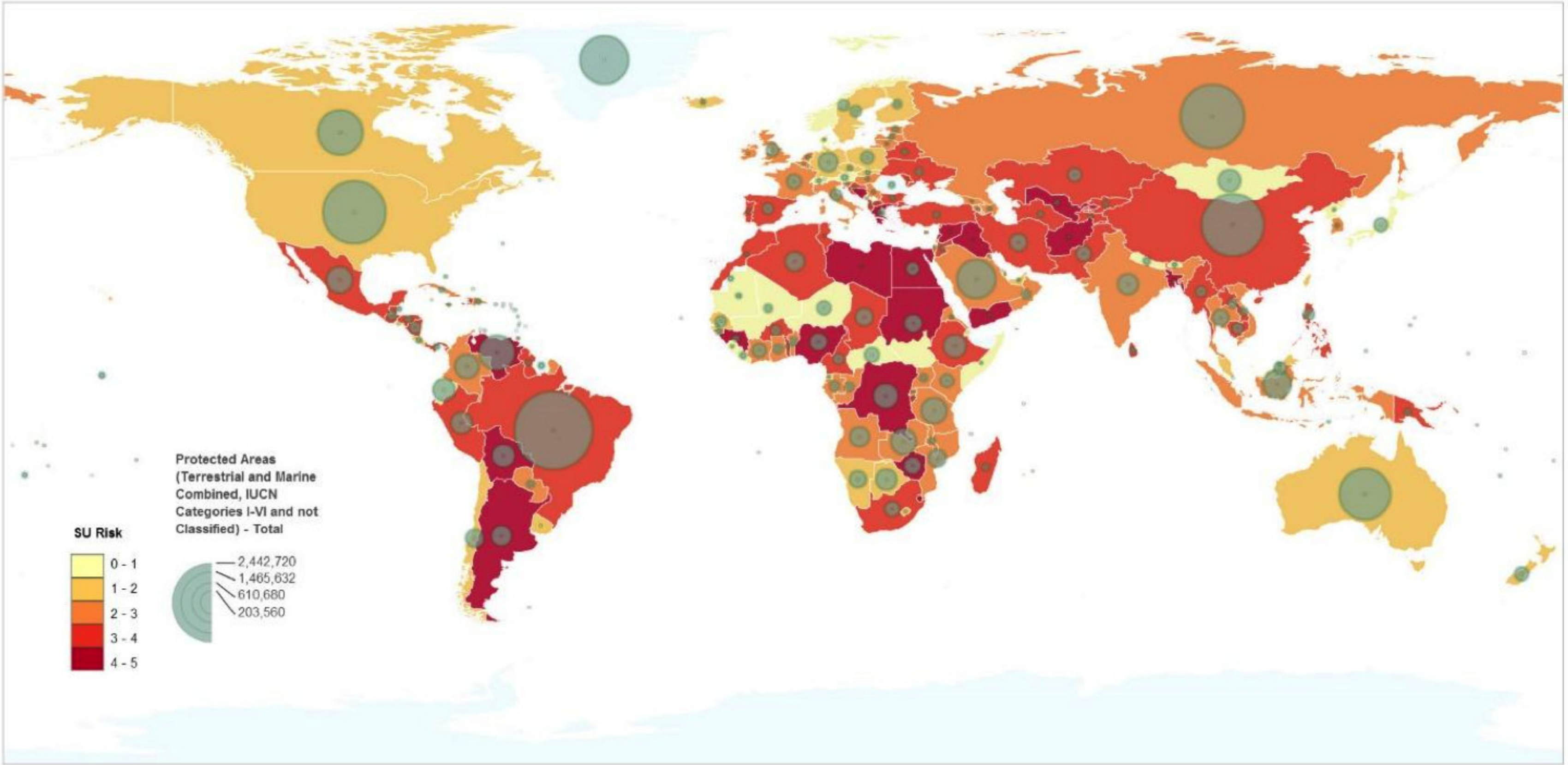


Figure 3.13: *Location of global protected areas (WDPA, 2015) within local conflict map indicated by social unrest index (Economist Intelligence Unit, 2014)*

Table 3.10: *Sources recording impacts of social unrest on protected areas*

Country	Protected Areas	Year	Source	Impacts
Vietnam	Dai Mui Nature Reserve	1961-1975	Dinh, 1984; Westing, 1996	Deforestation; Herbicide-induced mortality of wildlife
	Thanh Phu Nature Reserve			
	Vo Doi Nature Reserve			
	Lang Sen Nature Reserve			
Ethiopia	Lake Mburo National Park	1982-1983	Eltringham and Malpas, 1993	Decline in populations of elephants and large ungulates; Encroachment and degradation
	Gorilla National Park			
	Semliki Forest Reserve			
Rwanda	Parc National des Volcans	1990-1997	Plumptre et al., 1997	Decline in populations of gorillas and large mammals; Encroachment and degradation
	Parc National des Volcans		Ordway, 2015	
	Gishwati National Park			
	Mukura National Park			
Congo	Kahuzi-Beiga National Park	1990-1999	Hall et al., 1997	Decline in populations of elephants and large mammals; Encroachment and degradation; Burning of forests
Iraq	Tigris-Euphrates Marshlands	1999-2000	Partow, 2001	Drainage of marshlands
Nepal	Annapurna Conservation Area	2001-2003	Baral and Heinen, 2005	Poaching and illicit wildlife trade; Assaults on protected area staff
Sierra Leone	Gola Forest Reserve	1991-2001	Lindsell et al., 2010	Encroachment
Angola	Luiana Protected Reserve	1975-2012	Chase and Griffin, 2009	Decline in populations of elephants and large mammals
India	Nameri Tiger Reserve	1990-2013	Velho et al., 2014	Deforestation; increased instances of poaching
	Pakke Tiger Reserve			

Although the general pathways of social unrest and its eventual impact on protected areas appear to be broadly similar, the different social, political and even natural causes of these conflicts make it challenging to identify specific factors and pinpoint their respective impacts (Ortwin et al, 2012). Further empirical research and communication with protected area managers who have experience of parks entrenched in social conflicts is warranted for more comprehensive materiality mapping.

3.1.11.4 Manageability

Although the probability of social unrest appears to be increasing, the occurrences are dispersed in specific countries in Latin America, Africa, Europe and Central Asia. Moreover, the majority of these take the form of protests, and escalation into ochlocracy and uprisings are rare. Yet gaps exist in our understanding of the impacts these events have on protected areas which should be addressed.

Protected area management in areas at risk of social unrest should seek help from international aid agencies and NGOs while forming collaborations with experienced park management personnel to build resilience. Working together, protected area management in regions at risk of unrest will be able to share knowledge and develop best practices to cope with the impacts brought about by conflicts. This is especially crucial for protected areas located in countries at high risk but which have yet to experience conflict, as they probably lack the capability or capacity to manage these risks solely by themselves.

***Table 3.11:** Sources recording impacts of social unrest on protected areas*

	Protests	Civil Disorder	Ochlocracy	Uprising
Activities	Demonstrations Sit-ins Non-violent protests Vo Doi Nature Reserve	Riots Looting Arson	Systematic destruction aimed at targets of hatred	Civil War Sectarian violence
Destructiveness	Disruption to day-to-day activities No physical damage	Property directly targeted Cars damaged Arson	Systematic looting and destruction Specific groups being targeted Death and injury	Large scale physical and infrastructure damage High death toll Massacres
Functions affected	Economic and Development	Economic and Development Security issues	Economic and Development Government policies Security issues	Economic and Development Government policies Security issues
Dysfunctions	Inconvenient to travel	Lawlessness Unsafe to travel Restricted mobility	Resource supply shortage Lawlessness Unsafe to travel	Resource supply shortage Restricted mobility Lawlessness Unsafe to travel
Negative Impacts	Reduced visitorship	Reduced visitorship Unable to access funds	Instances of poaching and deforestation Encroachment Halt of tourism activities Unable to access funds	Widespread poaching and deforestation Encroachment Damage to infrastructure Halt of tourism activities Unable to access funds

Bibliography

Abrahams, D.J. (2008) Brand Risk: Adding risk literacy to brand management. Gower: USA.

Adams J.S. & McShane T.O. (1996) The Myth of Wild Africa: Conservation Without Illusion. University of California Press: USA.

Adams, W.M. (2004). Against extinction: the story of conservation. Earthscan, London, UK.

Agnew, D. J., Pearce, J., Pramod, G., Peatman, T., Watson, R., Beddington, J. R. & Pitcher, T. J. (2009). Estimating the Worldwide Extent of Illegal Fishing. PLoS ONE, 4, e4570.

Agnew, David J., John Pearce, Ganapathiraju Pramod, Tom Peatman, Reg Watson, John R. Beddington, and Tony J. Pitcher. "Estimating the worldwide extent of illegal fishing." PLoS one 4, no. 2 (2009): e4570.

Agrawal, A. (2012). Local institutions and the governance of forest commons. Comparative Environmental Politics: Theory, Practice, and Prospects, 313-340.

Aitken, S. N., Yeaman, S., Holliday, J. A., Wang, T. and Curtis-McLane, S. (2008), Adaptation, migration or extirpation: climate change outcomes for tree populations. Evolutionary Applications, 1, 95–111.

Alexander, N. (2014). The Emerging Multi-polar World Order: Its Unprecedented Consensus on a New Model for Financing Infrastructure Investment and Development (Heinrich Böll Foundation North America).

Alho, CJR. (2008). The value of biodiversity. Brazilian Journal of Biology, 68 (4), 1115-1118

Al Houty, W., Abdal, M., & Zaman, S. (1993). Preliminary assessment of the Gulf War on Kuwaiti desert eco system. Journal of Environmental Science & Health Part A, 28(8), 1705-1726.

All Africa (2013): Chinese investors in tourism. Available at: http://allafrica.com/stories/201307220 152.html [Accessed on 2 September 2014]

Allen T.M. (2000) Biodiversity: Exploring values and priorities in conservation. John Wiley & Sons, UK.

Alley, R., (2007) Summary for policymakers. Climate Change 2007: The Physical Science Basis, in S. Solomon et al, Eds., Cambridge University Press, 1–18.

Allison, H. E. & Hobbs, R. J. (2004). Resilience and adaptive capacity of the Western Australian agricultural region: A social-ecological system. Ecology and Society, 9, Article 3.

Alsdirawi, F., (1991) The negative impact of the Iraqi invasion on Kuwait's protected areas. Fourth World Congress on National Parks and Protected Areas.

Andrade, G. S. M., and J. R. Rhodes. 2012. Protected areas and local communities: an inevitable partnership toward successful conservation strategies? Ecology and Society 17(4): 14.

Andréassian V. (2004) Waters and forests: From historical controversy to scientific debate. Journal of Hydrology, 291, 1–27.

Anthony, K.R.N. & Fabricius, K.E.., (2008). Shifting roles of heterotrophy and autotrophy in coral energetics under varying turbidity. Journal of Experimental Marine Biology and Ecology, 252(2), 221-53.

Archabald, K. & Naughton-Treves, L. (2001). Tourism revenue-sharing around national parks in Western Uganda: early efforts to identify and reward local communities. Environmental conservation, 28, 135-149.

Armsworth, P.R. et al. (2007). Ecosystem-Service Science and the Way Forward for Conservation. Conservation Biology, 21, 1383–1384.

Asdak, C., Jarvis, P. G. & Gardingen, P. V. (1998). Evaporation of intercepted precipitation based on an energy balance in unlogged and logged forest areas of central Kalimantan, Indonesia. Agricultural and forest meteorology, 92, 173-180.

Askham, B. 2014. Local people may be the key to scaling back wildlife crime. ECOS, 25 November 2014. From ECOS online http://www.ecosmagazine.com/?paper=EC14284 [Accessed: 27 March 2015]

Atrium Ltd. (2003) Making sense of an uncertain world\Interband (2007) All brands are not created equal: Best Global Brands 2007, Interbrand in association with BusinessWeek

Austin, J.E. and Bruch, C.E. (2000). The Environmental Consequences of War: Legal, Economic, and Scientific Perspectives. Cambridge University Press.

Azfar, O., Lee, Y. & Swamy, A. (2001). The causes and consequences of corruption. Annals of the American Academy of Political and Social Science, 573, 42–56.

Azqueta D. & Sotelsek D. (2007). Valuing nature: From environmental impacts to natural capital. Ecological Economics, 63, 22–30.

Baker, A.C., Glynn, P.W. & Riegl, B., 2008. Climate change and coral reef bleaching: An ecological assessment of long-term impacts, recovery trends and future outlook, Estuarine Coastal and Shelf Science, 345, 229-36

Balmford, A. & Whitten, T. (2003). Who should pay for tropical conservation, and how could the costs be met? Oryx, 37, 238–250.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R.E., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, V., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J.R., Trumper, K., Turner, R.K. (2002). Economic reasons for conserving wild nature. Science, 297, 950-953.

Balmford, A., Gaston, K.J., Blyth, S., James, A., Kapos, V. (2003). Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. Proceedings of the National Academy of Sciences, 100, 1046–1050.

Balmford, A., Green, J. M. H., Anderson, M., Beresford, J., Huang, C., Naidoo, M., Manica, A. (2015) Walk on the wild side: estimating the global magnitude or visits to protected areas. PLOS Biology. 58605-58614.

Baral, N. and Heinen, J.T. (2005) The Maoist people's war and conservation in Nepal, Politics and the Life Sciences, 2005, 24(1): 2

Barber, C. V. & Talbott, K. (2003). The Chainsaw and the Gun: the role of the military in deforesting Indonesia. . In: Price, S. V. (ed.) War and Tropical Forests: Conservation in Areas of Armed Conflict.: The Haworth Press, Inc.

Barbosa, P.M., J.-M. Grégoire, and J. M. C. Pereira, (1997). Detection of burned areas in Africa using a multitemporal, multithreshold analysis of NOAA-AVHRR GAC data. Proceedings of European Symposium on Aerospace and Remote Sensing, 3222, pp. 67–75.

Barnes, R. F. W., Blom, A. & Alers, M. P. T. (1995). A review of the status of forest elephants Loxodonta africana in Central Africa. . Biological Conservation, 71, 125–132.

Barry D. & Oelschlaeger M. (1996). A science for survival: values and conservation biology. Conservation Biology, 10, 904–920.

Bavinck, M. et al. (2013). Conflict and cooperation over natural resources – Conceptual approaches. Boca Raton: CRC Press.

Bax, N., Williamson, A., Aquero, M., Gonzalez, E., Geeves, W., (2003). Marine invasive alien species: a threat to global biodiversity. Marine Policy, 27(4), 313–323

Beall, J., Goodfellow, T. and Rodgers, D. (2013). Cities and Conflict in Fragile States in the Developing World. Urban Studies 50:3065–3083.

Beamish, R. (2013) National and State Parks Suffering From Budget Cuts. The Pew Charitable Trust Website. Available on: http://www.pewtrusts.org/en/research-and-analysis/blogs/stateline/2013/05/24/national-and-state-parks-suffering-from-budget-cuts [Accessed 9 September 2015]

Benítez-López, A., Alkemade, R., and Verweij, P.A. (2010). The impacts of roads and other infrastructure on mammal and bird populations: a meta-analysis. Biological Conservation, 143, 1307–1316.

Bernard, E., Penna, L. & Araújo, E. (2014) Downgrading, Downsizing, Degazettement, and Reclassification of Protected Areas in Brazil. Conservation Biology, 28, 939-950.

Bernard, E., Penna, L. A. O., & Araújo, E. (2014). Downgrading, downsizing, degazettement, and reclassification of protected areas in Brazil. Conservation Biology, 28(4), 939-950.

Beyerlin, U. and Marauhn, T. (2011). International Environmental Law. Hart Publishing Ltd.

Bhattacharyya, S. & Hodler, R. (2010). Natural resources, democracy and corruption. European Economic Review, 54, 608-621..

Bijlsma, L. (1996). Coastal zones and small islands. In Watson, R.T., Zinyowera, M.C., Moss, R.H. (eds.) 1996. Climate Change 1995: Impacts, Adaptations and Mitigation of Climate Change: Scientific-Technical Analyses. Contribution of Working Group II to the Second Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge: Cambridge University Press.

Birdlife International (2012). A Comparative Study of Costs and Benefits of Soda Ash Mining and Promotion of Ecotourism and Sustainable Use of Natural Resources in Lake Natron Basin, Tanzania. BirdLife Africa Partnership Secretariat, Nairobi, Kenya

Blackman, A. (2015). Strict versus mixed-use protected areas: Guatemala's Maya Biosphere Reserve. Ecological Economics, 112, 14-24.

Blake, S. et al. (2007). Forest elephant crisis in the Congo Basin. PLoS Biol. 5, e111. 11. Adeney, J.M.,

Christensen, N., and Pimm, S.L. (2009). Reserves protect against deforestation fi res in the Amazon. PLoS 1, 4, e5014.

Boissière, M., D. Sheil, I. Basuki, M. Wan, and H. Le. (2009). Can engaging local people's interests reduce forest degradation in central Vietnam? Biodiversity Conservation, 18, 2743–2757.

Borrini-Feyerabend, G., N. Dudley, T. Jaeger, B. Lassen, N. Pathak Broome, A. Phillips and T. Sandwith (2013). Governance of Protected Areas: From understanding to action. Best Practice Protected Area Guidelines Series No. 20, Gland, Switzerland: IUCN. xvi + 124pp.

Bösze, Szilvia and Meyer, Hildegard. Editors (2014) Regional Development Opportunities of Protected Areas and Natural Assets in the Carpathians. Work Package 4. Integrated Management of Biological and Landscape Diversity for Sustainable Regional Development and Ecological Connectivity in the Carpathians. WWF Danube-Carpathian Programme, Vienna, Austria.

Bouwma. I.M., R. van Apeldoorn, D.A. Kamphorst, (2010). Current practices in solving multiple use issues of Natura 2000 sites: Conflict management strategies and participatory approaches. Alterra, Wageningen, the Netherlands.

Bovarnick, A., J. Fernandez B., Galindo, J. & Negret, H., (2010) Financial Sustainability of Protected Areas in Latin America and the Caribbean: Investment Policy Guidance, United Nations Development Programme (UNDP) and The Nature Conservancy (TNC).

Bowen-Jones, E. & Abigail, E. (2002). Identifying appropriate flagship species: the importance of culture and local contexts. Oryx, 36(2), 189-195.

Bowen-Jones, E. (2003). Bushmeat: Traditional Regulation or Adaptation to Market Forces. In The Trade in Wildlife: Regulation for Conservation. ed. Sarah Oldfield. Earthscan Publications Ltd. London.

Bown, N., Gray, T. S., & Stead, S. M. (2013). Contested Forms of Governance in Marine Protected Areas: A Study of Co-management and Adaptive Co-management. Routledge.

BP, (2015) Country insights: Brazil. Available at: http://www.bp.com/en/global/corporate/about-bp/energy-economics/energy-outlook/country-and-regional-insights/brazil-insights.html [Accessed: 2 July, 2015]

Braat, L.C. and de Groot, R. (2012). The ecosystem services agenda: bridging the worlds of natural science and economics, conservation and development, and public and private policy. Ecosystem Services, 1, 4–15.

Braun, B. & Castree, N. (2005). Remaking reality: nature at the millenium, Routledge.
Broadbent, E. N., Asner, G. P., Keller, M., Knapp, D. E., Oliveira, P. J. & Silva, J. N. (2008). Forest fragmentation and edge effects from deforestation and selective logging in the Brazilian Amazon. Biological conservation, 141, 1745-1757.

Brockington D. (2002) Fortress Conservation: The Preservation of the Mkomazi Game Reserve, Tanzania. James Currey Publishers: UK.

Brockington, D. & Igoe, J. (2006) Eviction for Conservation: A Global Overview. Conservation and Society. 4:424-470.

Brockington, D. (2004) Community conservation, inequality and injustice: Myths of power in protected area management. Conservation and society 2(2): 411.

Brooker, L., Brooker, M. & Cale, P. (1999). Animal dispersal in fragmented habitat: measuring habitat connectivity, corridor use and dispersal mortality. Conservation Ecology (online) (URL http://www.consecol.org/vol3/iss1/art4).

Brown, O., & McLeman, R. (2009). A recurring anarchy? The emergence of climate change as a threat to international peace and security: Analysis. Conflict, security & development, 9(3), 289-305.

Bruner A.G., Gullison R.E., & Balmford A. (2004) Financial Costs and Shortfalls of Managing and Expanding Protected-Area Systems in Developing Countries. BioScience, 54, 1119.

Brunner, R. (2002). Problems of governance. In R. Brunner, C. Colburn, C. Cromley, R. Klein, & E. Olson (Eds.), Finding common ground: Governance and natural resources in the American West (pp. 1–47). New Haven, CT: Yale University Press.

Brunner, R., Steelman, T. A., Coe-Juell, L., Cromley, C. M., Edwards, C. M., & Tucker, D. W. (Eds.). (2005). Adaptive governance: Integrating science, policy, and decision-making. New York, NY: Columbia University Press

Burgess et al, (2005). Major gaps in the distribution of protected areas for threatened and narrow range Afrotropical plants. Biodiversity and Conservation, 14 (8), 1877 – 1894.

Burgess, N., Kilahama, F., Nderumaki, M., Kahemela, A. (2005) Mining: Has the mining problem in the Eastern Arc mountains been solved? The Arc Journal. Tanzania Forest Conservation Group, biannual Newsletter Issue No.19. ISSN 0856-8715

Burgman, M. (2005). Risks and decisions for conservation and environmental management. Cambridge University Press: Cambridge.

Business in the Community. (2011) Business case guide for biodiversity and ecosystems services. Natural England. Available at : http://www.bitc.org.uk/sites/default/files/webform/files/business_case_guide_for_biodiversity_and_ecosystem_services_feb_2011.pdf [Accessed: 5 November 2011]

Butchart, S.H.M. et al. (2010). Global Biodiversity: Indicators of Recent Declines. Science, 328, 1164–1168.

Callister, D. J. (1999). Corrupt and illegal activities in the forestry sector: current understandings and implications for World Bank forestry policy. Forest Policy Implementation Review and Strategy Development: Analytical Studies.World Bank, Washington DC.

Carr, L. A., Stier, A. C., Fietz, K., Montero, I., Gallagher, A. J. & Bruno, J. F. (2013). Illegal shark fishing in the Galapagos Marine Reserve. Marine Policy, 39, 317-321.

Castree, N. (2008b). Neoliberalising nature: the logics of deregulation and reregulation. Environment and planning. A, 40, 131.

Castro, Gonzalo. (2003). Conservation Finance: The Long Road to Sustainability. Presentation to the Sustainable Finance Stream, Vth IUCN World Parks Congress, 8–17 September 2003: Durban, South Africa.

Chae, D., Wattage, Premachandra and Pascoe, S. (2011) Recreational benefits from a marine protected area: a travel cost analysis of Lundy. Tourism Management, 33 (4). pp. 971-977

Chase, M. J., & Griffin, C. R. (2009). Elephants caught in the middle: impacts of war, fences and people on elephant distribution and abundance in the Caprivi Strip, Namibia. African Journal of Ecology, 47(2), 223-233.

Chen, J.L., Wilson, C.R., Tapley, B.D., (2006). Satellite Gravity Measurements Confirm Accelerated Melting of Greenland Ice Sheet. Science ,313(5795), 1958-1960

Christoffersen, P., Hambrey, M.J., (2006). Is the Greenland ice sheet in a state of collapse? Geology Today, 22, 98-103.

Clark, M.E., K. A. Rose, D. A. Levine and W. W. Hargrove. (2001) Predicting Climate Change Effects on Appalachian Trout: Combining GIS and Individual-Based Modelling. Ecological Applications, 11(1), 161-178.

Cochrane, M. A. & Laurance, W. F. (2002). Fire as a large-scale edge effect in Amazonian forests. Journal of Tropical Ecology, 18, 311-325.

Cochrane, M.A., Ane Alencar, Mark D. Schulze, Carlos M. Souza Jr., Daniel C. Nepstad, Paul Lefebvre, and Eric A. Davidson. (1999). Positive feedbacks in the fire dynamic of closed canopy tropical forests. Science, 284 (5421), 1832-1835.

Collen, B., Loh, J., Whitmee, S., McRae, L., Amin, R., Baillie, J.E.M. (2009). Monitoring change in vertebrate abundance: the Living Planet Index. Conservation Biology, 23(2), 317-327.

Comparing protests (2014). The Economist. Available at: http://www.economist.com/blogs/graphicdetail/2014/09/daily-chart-24 [Accessed: 22 June 2015].

Constantine, R., Brunton, D. H., & Dennis, T. (2004). Dolphin-watching tour boats change bottlenose dolphin (Tursiops truncatus) behaviour. Biological Conservation, 117(3), 299-307.

Constanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S., Turner, R. K. (2014). Changes in the global value of ecosystem services. Global Environmental Change, 26, 152-158.

Contreras-Hermosilla, A. (2001). Forest law enforcement. , World Bank, Washington DC.

Cosier, S., (2007). War's effect on Iraq's environment. Mother Nature Network. Available on: http://www.mnn.com/search/google/keyword?query=iraq+war [Accessed 20 July 2015].

Costanza R. (2001). Visions, Values, Valuation, and the Need for an Ecological Economics. BioScience, 51, 459.

Cox, P.M., Betts, R.A., Collins, M., Harris, P., Huntingford, C., Jones, C.D., (2004). Amazon dieback under climate carbon cycle projections of the 21st century. Theoretical and Applied Climatology, 78(1-3), 137-156

Crocker T.D. (1999). A short history of environmental and resource economics. Handbook of Environmental and Resource Economics, 32–45.

Daily G. (1997) Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, USA.

Dale, V.H., Linda A. Joyce, Steve Mcnulty, Ronald P. Neilson, Matthew P. Ayres, Michael D. Flannigan, Paul J. Hanson, Lloyd C. Irland, Ariel E. Lugo, Chris J. Peterson, Daniel Simberloff, Frederick J. Swanson, Brian J. Stocks, and B. Michael Wotton. (2001). Climate change and forest disturbances. BioScience, 51(9), 723-734.

Daly, M. (2008). Bush forest budget cut called disastrous. USA Today Online. Available from www.usatoday.com [Accessed 21 July 2015].

Damania, R., Fredriksson, P. G. & List, J. A. (2003). Trade liberalization, corruption, and environmental policy formation: theory and evidence. Journal of environmental economics and management, 46, 490-512.

Davies, D.K., Shriram, I., Wong. M.M., and Christopher, O. J., (2009). Fire Information for Resource Management System: Archiving and Distributing MODIS Active Fire Data. IEE Transactions on geoscience and remote sensing, 47(1).

Davies Barnard, T., Valdes, P., Singarayer, J., Wiltshire, A. & Jones, C. (2015). Quantifying the relative importance of land cover change from climate and land use in the representative concentration pathways. Global Biogeochemical Cycles.

Davis, M.B., Shaw, R.G., Etterson, J.R., (2005). Evolutionary responses to changing climate. Ecology, 86(7), 1704-1714.

Debonnet, G. and Hillman-Smith, K. (2004). Supporting protected areas in a time of political turmoil: the case of World Heritage Sites in the Democratic Republic of Congo. Parks. 14(1): 9–16.

Debonnet, G., & Hillman-Smith, K. (2004). Supporting protected areas in a time of political turmoil: the case of World Heritage Sites in the Democratic Republic of Congo. Parks, 14, 9-16.

DECC (2007) Department of Environment and Climate Change. BioBanking: Biodiversity Banking and Offsets Scheme – Scheme Overview. Sydney, Australia

Díaz S., Demissew S., Carabias J., Joly C., Lonsdale M., Ash N., Larigauderie A., Adhikari J.R., Arico S., Báldi A., Bartuska A., Baste I.A., Bilgin A., Brondizio E., Chan K.M., Figueroa V.E., Duraiappah A., Fischer M., Hill R., Koetz T., Leadley P., Lyver P., Mace G.M., Martin-Lopez B., Okumura M., Pacheco D., Pascual U., Pérez E.S., Reyers B., Roth E., Saito O., Scholes R.J., Sharma N., Tallis H., Thaman R., Watson R., Yahara T., Hamid Z.A., Akosim C., Al-Hafedh Y., Allahverdiyev R., Amankwah E., Asah T.S., Asfaw Z., Bartus G., Brooks A.L., Caillaux J., Dalle G., Darnaedi D., Driver A., Erpul G., Escobar-Eyzaguirre P., Failler P., Fouda A.M.M., Fu B., Gundimeda H., Hashimoto S., Homer F., Lavorel S., Lichtenstein G., Mala W.A., Mandivenyi W., Matczak P., Mbizvo C., Mehrdadi M., Metzger J.P., Mikissa J.B., Moller H., Mooney H. a, Mumby P., Nagendra H., Nesshover C., Oteng-Yeboah A.A., Pataki G., Roué M., Rubis J., Schultz M., Smith P., Sumaila R., Takeuchi K., Thomas S., Verma M., Yeo-Chang Y., & Zlatanova D. (2015) The IPBES Conceptual Framework — connecting nature and people. Current Opinion in Environmental Sustainability, 14, 1–16.

Dinh, H. (1984). Long-term changes in dense inland forest following herbicidal attack. In Westing, A.H. eds. (1984), Herbicides in war: the long term ecological and human consequences. Stockholm International Peace Research Institute. Taylor& Francis, London.

Dirzo, R. & Raven, P. H. (2003). Global state of biodiversity and loss. Annual Review of Environment and Resources, 28, 137-167.

DPAW (2015) Available at: http://www.dpaw.wa.gov.au/management/fire [Accessed: 20 March 2015)

DPI, (2015) PRODES. Available at: http://www.dpi.inpe.br/prodesdigital/prodesuc.php [Accessed: 4 May 2015]

Draulans, D. and Van Krunkelsven, E. (2002) The impact of war on forest areas in the Democratic Republic of Congo. Oryx 6(1): 35-40

Dudley, J. P., Ginsberg, J. R., Plumptre, A. J., Hart, J. A., & Campos, L. C. (2002). Effects of war and civil strife on wildlife and wildlife habitats. Conservation Biology, 16(2), 319-329.

Dudley, N. (ed.) (2008) Guidelines for Appling Protected Areas Management Categories. IUCN: Gland, Switzerland. p.8-9. Web link

Dudley, N., Belokurov, A., Higgins-Zogib, L., Hockings, M., Stolton, S., & Burgess, N. (2007). Tracking progress in managing protected areas around the world: an analysis of two applications of the management effectiveness tracking tool developed by WWF and the World Bank. WWF International.

Dudley, N., Groves, C., Redford, K. H., & Stolton, S. (2014). Where now for protected areas? Setting the stage for the 2014 World Parks Congress. Oryx, 48(04), 496-503.

Duffy, Rosaleen, and Lorraine Moore. (2010). Neoliberalising Nature? Elephant back tourism in Thailand and Botswana. Antipode, 42(3), 742-766.

Eagles, P. F. (2004). Trends affecting tourism in protected areas. In Policies, methods and tools for visitor management. Proceedings of the Second International Conference on Monitoring and Management of Visitor Flows in Recreational and Protected Areas.

Edwards, D. P., Sloan, S., Weng, L., Dirks, P., Sayer, J. and Laurance, W. F. (2014), Mining and the African environment. Conservation Letters, 7, 302–311.

Ehrlich, P.R. and Ehrlich, A.H. (1981). Extinction: The Causes and Consequences of the Disappearance of Species. Random House.

Ellis, E. C. (2013). Sustaining biodiversity and people in the world's anthropogenic biomes. Current Opinion in Environmental Sustainability, 5, 368-372.

Ellison, J.C. (2000). How South Pacific mangroves may respond to predicted climate change and sea- level rise. Chapter 15, pages 289-301 In A. Gillespie and W. Burns, (eds.) Climate Change in the South Pacific: Impacts and Responses in Australia, New Zealand, and Small Islands States, Kluwer Academic Publishers, Dordrecht, pp. 289-301.

Eltringham, S. K., & Malpas, R. C. (1993). The conservation status of Uganda's game and forest reserves in 1982 and 1983. African Journal of Ecology, 31(2), 91-105.

Emerton L., Bishop J., & Thomas L. (2006) Sustainable financing of protected areas. A global review of challenges and options. IUCN, Gland, Switzerland.

EPA (1995) Environmental Protection Agency. Federal Guidance for the Establishment, Use and Operation of Mitigation Banks. Federal Register: November 28, 1995 (Volume 60, Number 228)

Estrada, G.C.D., Soares, M.L.G., Fernadez, V., de Almeida, P.M.M. (2015) The economic evaluation of carbon storage and sequestration as ecosystem services of mangroves: a case study from southeastern Brazil. International Journal of Biodiversity Science, Ecosystem Services & Management, 11(1), 29-3

European Union (EU) Biodiversity Strategy (2020) Our life insurance, our natural capital: an EU biodiversity strategy to 2020 (2011/2307 (INI). European Parliament resolution of 20 April 2012

Eva, H. and E. F. Lambin, (1998). Remote sensing of biomass burning in tropical regions: Sampling issues and multisensor approach. Remote Sensing and the Environment, 64(3), 292–315, 1998.

Fearnside, P. M. (2002). Greenhouse gas emissions from a hydroelectric reservoir (Brazil's Tucuruí Dam) and the energy policy implications. Water, Air, and Soil Pollution, 133, (1-4), 69-96.

Fearnside, P.M. and Graca, P.M.L.d. (2006). BR-319: Brazil's Manaus-Porto Velho highway and the potential impact of linking the arc of deforestation to central Amazonia. Environmental Management. 38, 705–716.

Ferraro, P. (2005). Corruption and conservation: the need for empirical analyses. A response to Smith & Walpole. Oryx, 39, 257-259.

Festa-Biancher, M., & Apollonio, M. (2003) Animal Behavior and Wildlife Conservation. Island Press Finer M., Jenkins C.N., Pimm S.L., Keane B., & Ross C. (2008). Oil and gas projects in the Western Amazon: threats to wilderness, biodiversity, and indigenous peoples. PloS one, 3, e2932.

Fjeldså et al. (2004). Where are the major gaps in the reserve network for Africa's mammals? Oryx, 38 (1), 17-25.

Flores, M., Rivero, G., León, F., Chan, G. et al, (2008), Financial Planning for National Systems of Protected Areas: Guidelines and Early Lessons. The Nature Conservancy, Arlington, Virginia, US.

Foellmi, R. & Oechslin, M. (2007). Who gains from non-collusive corruption? Journal of Development Economics, 82, 95-119.

Folke, C. (2006). The economic perspective: conservation against development versus conservation for development. Conservation Biology, 20, 686–688.

Forrest, J. L., Mascia, M. B., Pailler, S., Abidin, S. Z., Araujo, M. D., Krithivasan, R., & Riveros, J. C. (2015). Tropical deforestation and carbon emissions from protected area downgrading, downsizing, and degazettement (PADDD). Conservation Letters, 8(3), 153-161.

Foster, S. E. (2002). Aristotle and the Environment. Environmental Ethics,24(4), 409-428.

Friends of Serengeti (2015). Uranium mine. Available at: http://friendsofserengeti.org/conservation-news/tanzania-to-allow-uranium-mining-in-the-selous-game-reserve/ [Accessed 3 September 2015]

Funbio, (2015). Fundo Brasileiro Para A Biodiversidade Website. Available at: http://www.funbio.org.br/ [Accessed: 7 August 2015]

Gantioler, S., Brink, P., Bassi, S., Kettunen, M., McConville, A., & Rayment, M. (2010). Financing Natura 2000–Financing needs and socio-economic benefits resulting from investment in the network. IEEP, Brussels.

Garrett, K. A., S. P. Dendy, E. E. Frank, M. N. Rouse, and S. E. Travers. (2006). Climate Change Effects on Plant Disease: Genomes to Ecosystems. Annual Review of Phytopathology, 44, 489-509.

GEF (2009) Project ID 3253. Strengthening the Protected Area Network in Southern Tanzania: Improving the Effectiveness of National Parks in Addressing Threats to Biodiversity

GEF (2011). Strengthening the Protected Area Network in Southern Tanzania: Improving the effectiveness of national parks in addressing threats to biodiversity. Project Document (ID 3965). http://www.tz.undp.org/content/dam/tanzania/Strengthening%20the%20protected%20area%20network%20in%20Southern%20Tanzania.pdf

Geisler, C., (2003). A new kind of trouble: evictions in Eden. International Social Science Journal. 55:175.

Gentry, A. H. (1992). Tropical forest biodiversity: distributional patterns and their conservational significance. Oikos, 19-28.

German, T., & Rande, J. (1998). The Reality of Aid 1998/1999: an independent review of poverty reduction and development assistance, 1998/1999. Earthscan, London.

Gillson, L., Ladle, R. J. & Araujo, M. B. (2011). Baselines, Patterns and Process. In: LADLE, R. J. & WHITTAKER, R. J. (eds.) Conservation Biogeography. Blackwell-Wiley Publishing, Ltd.

Gomez-Baggethun E. & Ruiz-Perez M. (2011). Economic valuation and the commodification of ecosystem services. Progress in Physical Geography, 35, 613–628.

Gómez-Baggethun E., de Groot R., Lomas P.L., & Montes C. (2010). The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. Ecological Economics, 69, 1209–1218

Goodale, U.M. at el. (2003) Transboundary Protected Areas: The Viability of Regional Conservation Strategies. Binghamton, NY: Food Products Press, 2003.

Goss, J. R. and G. S. Cumming (2013). Networks of wildlife translocations in developing countries: an emerging conservation issue? Frontiers in Ecology and the Environment 11(5): 243-250.

Grants on the Web (2015). Available at: http://gotw.nerc.ac.uk/goti.asp?c=1 [Accessed: 3 September 2015]

Green, J. M. H. et al. (2012). Estimating management costs of protected areas: a novel approach from the Eastern Arc Mountains, Tanzania. Biological Conservation, 150, 5–14.

Green, R. E., Cornell, S. J., Scharlemann, J. P., & Balmford, A. (2005). Farming and the fate of wild nature. Science, 307(5709), 550-555.

Groombridge, B. (1992). ed., Global Biodiversity: Status of the Earth's Living Resources. London: Chapman & Hall

Haddad, N. M., Brudvig, L. A., Clobert, J., Davies, K. F., Gonzalez, A., Holt, R. D. & Townshend, J. R. (2015). Habitat fragmentation and its lasting impact on Earth's ecosystems. Science Advances, 1(2), e1500052.

Haedrich, R. L. & Hamilton, L. C. (2000). The fall and future of Newfoundland's cod fishery. Society & Natural Resources, 13, 359-372.

Hagel, C. T. (2014). Quadrennial Defense Review. Washington, DC: Department of Defense, March, 4, 2014.

Haggart, M. (2009) Proposal to mine aspiring. Otago Daily Times. December 1, 2009. Available from: http://www.odt. co.nz/news/national/84055/proposal-mine-aspiring. [Accessed: 20 May 2015.]

Haila, Y. & Kouki, J. (1994). The phenomenon of biodiversity in conservation biology. Annales Zoologici Fennici, 31: 5-18.

Haldén, P. (2007). The geopolitics of climate change. Challenges to the International System. Stockholm: FOI.

Hall, J., Inogwabini, B. I., Williamson, E., Omari, I., Sikubwabo, C., & White, L. (1997). A survey of elephants (Loxodonta africana) in the Kahuzi Biega National Park lowland sector and adjacent forest in eastern Zaire. African Journal of Ecology, 35(3), 213-223.

Hance, J. (2011). Goodbye national parks: when 'eternal' protected areas come under attack. Available on:http://news.mongabay.com/2011/03/goodbye-national-parks-when-eternal-protected-areas-come-under-attack/. [Accessed: 18 September 2015].

Hanks, J. & Attwell, C. A. M. (2003). Financing Africa's Protected Areas. Report from the 5th World Parks Congress: Sustainable Finance Stream. Durban, South Africa.

Hannah, L., G. F. Midgley and D. Millar (2002). Climate Change-Integrated Conservation Strategies. Global Ecology and Biogeography. 485-495

Hansen, A. J., & DeFries, R. (2007). Ecological mechanisms linking protected areas to surrounding lands. Ecological Applications, 17(4), 974-988.

Hansen, A. J., R. P. Neilson, V. H. Dale, C. H. Flather, L. R. Iverson, D. J. Currie, S. Shafer, R. Cook, and P. J. Bartlein. (2001). Global change in forests: Responses of species, communities, and biomes. BioScience, 51, 765–779.

Hanson, T. et al. (2009). Warfare in biodiversity hotspots. Conservation Biology 23:578–587.

Haugen, J., (2009) How travel warnings are hurting Honduras, and why you should think about ignoring them. Matador Network. Available from: http://matadornetwork.com/abroad/should-you-listen-to-travel-warnings-about-honduras/ [Accessed: 24 July 2015].

Heike, B., Elizabeth, D., Alex, V. & Markus, W. (2011) The effects of oil companies' activities on the environment, health and development in Sub-Saharan Africa. Available at: www.chathamhouse.org.uk/files/17180_250810gary.pdf [Accessed: 3 November 2015].

Helga, R. (2013) The political ecology and economy of protected areas and violence: a case study of the conflict of the Kivus in the Democratic Republic of Congo. PhD thesis, The London School of Economics and Political Science (LSE).

Helliwell, D.R. (1969). Valuation of Wildlife Resources. Regional Studies, 3, 41-49.

Henderson, J. V. & Kuncoro, A. (2004). Corruption in Indonesia. National Bureau of Economic Research.

Hicks, R. L., Parks, B. C., Roberts, J. T., & Tierney, M. J. (2008). Greening aid?: Understanding the environmental impact of development assistance. Oxford University Press.

Hobbs, R. J., Higgs, E. & Harris, J. A. (2009). Novel ecosystems: implications for conservation and restoration. Trends in Ecology & Evolution, 24, 599-605.

Hockings, M., Ervin, J., & Vincent, G. (2004). Assessing the management of protected areas: the work of the world parks congress before and after Durban. Journal of International Wildlife Law and Policy, 7(1-2), 31-42.

Hoegh-Guldberg, O. (1999). Climate change, coral bleaching and the future of the world's coral reefs. Marine and Freshwater Research, 50(8), 839-866.

Hoegh-Guldberg, O., P. J. Mumby, A. J. Hooten, R. S. Steneck, P. Greenfield, E. Gomez, C. D. Harvell, P. F. Sale, A. J. Edwards, K. Caldeira, N. Knowlton, C. M. Eakin, R. Iglesias-Prieto, N. Muthiga, R. H. Bradbury, A. Dubi, M. E. Hatziolos, (2007). Coral Reefs Under Rapid Climate Change and Ocean Acidification. Coral reefs, 318(5857), 1737-1742.

Hole, D. G., Willis, S. G., Pain, D. J., Fishpool, L. D., Butchart, S. H. M., Collingham, Y. C., Rahbek, C. and Huntley, B., (2009). Projected impacts of climate change on a continent-wide protected area network. Ecology Letters, 12, 420-431.

Honnay, O., Verheyen, K., Butaye, J., Jacquemyn, H., Bossuyt, B. and Hermy, M. (2002), Possible effects of habitat fragmentation and climate change on the range of forest plant species. Ecology Letters, 5, 525–530.

Hooper et al, (2012). A global synthesis reveals biodiversity loss as a major driver of ecosystem change. Nature, 486,105-108

Hooper, D. U. et al. (2005). Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecology Monograph, 75, 3-35.

Hooper, D.U., Chapin III, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, S., Naeem, B., Schmid, B., Setala, H., Symstad, A.J., Vandermeer, J., Wardle, D.A. (2005). Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecological Monographs, 75, 3-35

Houghton, J. (2009). Global Warming, Fourth Edition. Cambridge University Press.

Huber, O. (2001). Conservation and environmental concerns in the Venezuelan Amazon. Biodiversity & Conservation, 10, 1627-1643.

Huggett, A. J. (2005). The concept and utility of 'ecological thresholds' in biodiversity conservation. Biological Conservation, 124, 301-310.

Huntingford, C., Fisher, R. A., Mercado, L., Booth, B. B. B., Sitch, S., Harris, P. P., Cox, P. M., Jones, C. D., Betts, R. A., Malhi, Y., Harris, G. R., Collins, M. & Moorcroft, P. (2008). Towards quantifying uncertainty in predictions of Amazon 'dieback'. Philosophical Transactions of the Royal Society of London B: Biological Sciences, 363, 1857-1864.

ICMBio, (2015). Relação de ucs com plano de manejo. Available at: http://www.icmbio.gov.br/portal/biodiversidade /unidades-de-conservacao/planos-de-manejo/lista-plano-de-manejo.html [Accessed: 3 July 2015]

IEA, (2015). Oil market report: 13 October 2015. Available at: http://www.iea.org/oilmarketreport/omrpublic/currentreport/ [Accessed: 3 November 2015]

Igoe, J. & Croucher, B. (2007) Conservation, Commerce, and Communities: The story of community-based wildlife, Conservation & Society, 5 (4), 534-561

Infield, M. (2001) Cultural Values: A forgotten strategy for building community support for protected areas in Africa. Conservation Biology, 15(3), 800-802.

IPP Media (2014): Chinese investment in southern circuit infrastructure. Available at: http://www.ippmedia.com/frontend/index.php?l=73422 [Accessed: 3 September 2015]

IUCN (2000) Financing Protected Areas : guidelines for protected area managers. IUCN.

IUCN WCPA Task Force (1998). Task Force on Economic Benefits of Protected Areas of the World Commussion on Protected Areas (WCPA) of IUCN, in collaboration with the Economics Service Unit of IUCN. Economic Values of Protected Areas: Guidelines for Protected Area Managers. IUCN, Gland, Switzerland and Cambridge, UK. Xii+52pp

IUCN. (2009). Big Game Hunting in West Africa. What Is Its Contribution to Conservation? International Union for Conservation, Gland, Switzerland and Cambridge, UK.

Jachmann, H. (2008). Monitoring law-enforcement performance in nine protected areas in Ghana. Biological Conservation, 141(1), 89-99.

Jachmann, H.(2003). Elephant Poaching and Resource Allocation for Law Enforcement. In The Trade in Wildlife: Regulation for Conservation. ed. Sarah Oldfield. Earthscan Publications, London.

Jacobs, M.J. & Schloeder, C.A. (2001). Impacts of Conflict on Biodiversity and Protected Areas in Ethiopia. Washington, D.C.: Biodiversity Support Program.

Jacoby K. (2003) Crimes Against Nature: Squatters, Poachers, Thieves, and the Hidden History of American Conservation. University of California Press: USA.

James & Paine (1999). Global review of Protected Area Budgets and Staff – WCMC Biodiversity Series No. 10

James, A. N., Gaston, K. J., & Balmford, A. (1999). Balancing the Earth's accounts. Nature, 401(6751), 323-324.

James, A., Gaston, K. J., & Balmford, A. (2001). Can we afford to conserve biodiversity?. BioScience, 51(1), 43-52.

Jax, K. and Heink, U. (2015). Searching for the place of biodiversity in the ecosystem services discourse. Biological Conservation, 191, 198-205.

Jepson, P. (2015) Nature Conservation, in the AAG-Wiley International Encyclopedia of Geography: People, the Earth, Environment, and Technology encyclopedia@aag.org

Jepson, P. & Whittaker, R.J. (2002). Ecoregions in context: a critique with special reference to Indonesia. Conservation Biology, 16, 42-57.

Jepson, P., Jarvie, J. K., Mackinnon, K. & Monk, K. A. (2001). The end for Indonesia's lowland forests? Science, 292, 859-861.

Jeschke, J. M. and Strayer, D. L. (2008), Usefulness of bioclimatic models for studying climate change and Invasive Species. Annals of the New York Academy of Sciences, 1134, 1–24.

Jha, S. & Bawa, K. S. (2006). Population growth, human development, and deforestation in biodiversity hotspots. Conservation Biology, 20, 906-912.

Jimenez, J.A., A.E. Lugo, and G. Cintron. (1985). Tree mortality in mangrove forests. Biotropica, 17(3), 177-185

Johns, D. M. (2003). Growth, conservation, and the necessity of new alliances. Conservation Biology, 17, 1229-1237.

Johnston, M. E. (2006). Impacts of global environmental change on tourism in the polar regions. Tourism and global environmental change: Ecological, social, economic and political interrelationships, 37.

Joly, M., Bertrand, P., Gbangou, R.Y., White, M., Dubé J., & Lavoie, C. (2011). Paving the way for invasive species: Road type and the spread of common ragweed (Ambrosia artemisiifolia). Environmental Management, 48(3), 514-522.

Jones-Walters, L. and Mulder, I. (2009). Valuing nature: The economics of biodiversity, Journal for Nature Conservation, 17: 245-247

Joonmoo L., (1996). Poachers, Tigers, and Bears...Oh My--Asia's Illegal Wildlife Trade. Northwestern Journal of International Law & Business, 16, 497.

Jurskis, V., B. Bridges, P. de Mar. (2003). Fire management in Australia: the lessons of 200 years. Proceedings of the Joint Australia and New Zealand Institute of Forestry Conference, 27 April–1 May 2003, Ministry of Agriculture and Forestry, Wellington/Queenstown, New Zealand, 353–368.

Kallis G., Gómez-Baggethun E., & Zografos C. (2013). To value or not to value? That is not the question. Ecological Economics, 94, 97–105.

Kareiva, P., Marvier, M., Lalasz, R. (2012). Conservation in the Anthropocene: Beyond solitude and fragility. http://thebreakthrough.org/index.php/journal/past-issues/issue-2/conservation-in-the-anthropocene. [Accessed: 25 July 2015].

Karesh WB, Cook RA, Bennett EL, Newcomb J. (2005). Wildlife trade and global disease emergence. Emerging Infectious Diseases, 11, 1000–1002

Katz, T., Yahel, G., Yahel, R., Tunnicliffe, V., Herut, B., Snelgrove, P., Crusius, J. & Lazar, B. (2009). Groundfish overfishing, diatom decline, and the marine silica cycle: Lessons from Saanich Inlet, Canada, and the Baltic Sea cod crash. Global Biogeochemical Cycles, 23.

Kelly, R. P., Erickson, A. L., Mease, L. A., Battista, W., Kittinger, J. N., & Fujita, R. (2015). Embracing thresholds for better environmental management. Philosophical Transactions of the Royal Society of London B: Biological Sciences, 370(1659), 20130276.

Kideghesho J.R. (2006) Wildlife conservation and local land use conflicts in Western Serengeti, Tanzania. PhD Thesis. NTNU-trykk: Norway.

King, L.M. (2011) Investigating the role of the World Heritage brand in attracting visitors to protected areas in Queensland, Australia. PhD thesis, James Cook University.

Kinnaird, M. F., Sanderson, E. W., O'brien, T. G., Wibisono, H. T. & Woolmer, G. (2003). Deforestation trends in a tropical landscape and implications for endangered large mammals. Conservation Biology, 17, 245-257.

Krauss, C., & Reed, S. (2015) Shell exits Arctic as slump in oil prices forces industry to retrench. The New York Times. Available on: http://www.nytimes.com/2015/09/29/business/international/royal-dutch-shell-alaska-oil-exploration-halt.html?_r=0 [Accessed: 3 November 2015]

Klerk et al. (2004). Gaps in the protected area network for threatened Afrotropical birds. Biological Conservation 117 (5): 529-537

Kolstad, I. & Wiig, A. (2009). Is transparency the key to reducing corruption in resource-rich countries? World Development, 37, 521-532.

Kothari A., Suri S., & Singh N. (1995) Conservation in India: a new direction. Economic and Political Weekly, 2755–2766.

Kroner, R.E.G. (2015). Protected Areas and PADDD in Cambodia. Available on: http://rachelegolden.com/2015/05/29/protected-areas-and-paddd-in-cambodia/ [Accessed: 9 September 2015]

Lack, M. & Sant, G. (2008). Illegal, unreported and unregulated shark catch: A review of current knowledge and action.

Ladle R.J. & Whittaker R.J. (2011). Conservation Biogeography. John Wiley & Sons, UK.

Ladle, R.J., Bragagnolo, C., Gama, G.M., Malhado, A.C.M., Root-Bernstein, M. & Jepson. P., (2014). Private protected areas: Three key challenges. Environmental Conservation, 41, 239-240.

Lamb, W. (2013). Commentary on economic valuation of biodiversity. Ecological Economics, 89, 170-173.

Lang, C. (2014). Money Logging: Corruption, power and the destruction of Sarawak's forests. Ecologist. Available at: http://www.theecologist.org/reviews/2640306/money_logging_corruption_power_and_the_destruction_of_sarawaks_forests.html [Accessed: 4 November 2015].

Lasgorceix, A. & Kothari, A. (2009). Displacement and relocation of protected areas: a synthesis and analysis of case studies. Economic and Political Weekly. 49(XLIV): 37-47.

Lasswell, H. D. (1971). A Preview of Policy Sciences. New York, NY: American Elsevier.

Laurance, W. F. (2004). The perils of payoff: corruption as a threat to global biodiversity. Trends in Ecology & Evolution, 19, 399-401.

Laurance, W. F. (2013). Does research help to safeguard protected areas? Trends in Ecology & Evolution 28(5): 261-266.

Laurance, W. F., Cochrane, M. A., Bergen, S., Fearnside, P. M., Delamônica, P., Barber, C. & Fernandes, T. (2001). The future of the Brazilian Amazon. Science, 291(5503), 438-439.

Laurance, W. F., Peletier-Jellema, A., Geenen, B., Koster, H., Verweij, P., Van Dijck, P & Van Kuijk, M. (2015). Reducing the global environmental impacts of rapid infrastructure expansion. Current Biology, 25(7), R259-R262.

Laurance, W.F., Clements, G.R., Sloan, S., O'Connell, C., Mueller, N.D., Goosem, M., Venter, O., Edwards, D.P., Phalan, B., Balmford, A., Van Der Ree, R., and Arrea, I.B. (2014). A global strategy for road building. Nature, 513, 229–232.

Laurance, W.F., Goosem, M., and Laurance, S.G. (2010). Impacts of roads and linear clearings on tropical forests. Trends in Ecology and Evolution, 24, 659–669

Lausche, Barbara. (2011). Guidelines for Protected Areas Legislation. IUCN, Gland, Switzerland. xxvi + 370 pp.

Lawson, K. and Vines, A. (2014). Global Impacts of the Illegal Wildlife Trade The Costs of Crime, Insecurity and Institutional Erosion. Chatham House

Le Saout, S., Hoffmann, M., Shi, Y., Hughes, A., Bernard, C., Brooks, T.M., Bertzky, B., Butchart, S.H.M., Stuart, S.N., Badman, T., Rodrigues A.S.L. (2013) Protected areas and effective biodiversity conservation. Science, 342, 803-805.

Leader Williams, N. & Milner Gulland, E. (1993). Policies for the enforcement of wildlife laws: the balance between detection and penalties in Luangwa Valley, Zambia. Conservation Biology, 7, 611-617.

Leader-Williams, N. (1992). The World Trade in Rhino Horn: A review, TRAFFIC International, Cambridge.

Leader-Williams, N. (2003). Regulation and Protection: Successes and Failures in Rhnoceros Conservation. In The Trade in Wildlife: Regulation for Conservation ed. Sara Oldfield. Earthscan Publications Ltd. London.

Leader-Williams, N., Albon, S. & Berry, P. (1990). Illegal exploitation of black rhinoceros and elephant popu lations: patterns of decline, law enforcement and patrol effort in Luangwa Valley, Zambia. Journal of Applied Ecology, 1055-1087.

Leader-Williams, N., Kayera, J.A. Overton, G.L. (Eds.) (1996). Community-based Conservation in Tanzania, IUCN Gland, Switzerland and Cambridge, UK. ix + 226pp

Leishman, M. R., Masters, G. J., Clarke, I. P. and Brown, V. K. (2000), Seed bank dynamics: the role of fungal pathogens and climate change. Functional Ecology, 14, 293–299.

Lele, S., Wilshusen, P., Brockington, D., Seidler, R., & Bawa, K. (2010). Beyond exclusion: alternative ap proaches to biodiversity conservation in the developing tropics. Current Opinion in Environmental Sustainability. 2(1): 94-100.

Lemieux, A and Clarke, R. (2009). The international ban on ivory sales and its effects on elephant poaching in Africa. British Joural of Criminology, 49(4), 451-71

Lenton, T. M., Held, H., Kriegler, E., Hall, J. W., Lucht, W., Rahmstorf, S. & Schellnhuber, H. J. (2008). Tip ping elements in the Earth's climate system. Proceedings of the National Academy of Sciences, 105, 1786-1793.

Leverington F., Hockings M., & Costa K.L. (2010) Management effectiveness evaluation in protected areas – a global study. 1-70.

Leverington, F., Costa, K.L., Courrau, J., Pavese, H., Nolte, C., Marr, M., Coad, L., Burgess, N., Bomhard, B., Hockings, M. (2010) Management effectiveness evaluation in protected areas – a global study. Second edition 2010. The University of Queensland, Brisbane, Australia.

Lewis D., Kaweche G.B., & Mwenya A. (1990) Wildlife conservation outside protected areas - lessons from an experiment in Zambia. Conservation Biology, 4, 171-180.

Lindermann, J. (2007) Brand valuation: The economy of brands, Interbrand

Lindsell, J. A., Klop, E., & Siaka, A. M. (2011). The impact of civil war on forest wildlife in West Africa: mammals in Gola Forest, Sierra Leone. Oryx, 45(01), 69-77.

Lindsey, P.A., Balme, G.A., Booth, V.R., Midlane, N. (2012) The significance of African lions for the financial viability of trophy hunting and the maintenance of wild land. PLoS ONE, 7 (1): e29332.

Lips, K.R. et. al (2006) Emerging infectious disease and the loss of biodiversity in a Neotropical amphibian community. BioScience, 103(9), 3165-3170

Loarie, S.R., Philip B. Duffy, P.B., Hamilton, H., Asner, G.P., Field, C.B., & Ackerly, D.D. (2009). The velocity of climate change. Nature, 462, 1052-1055.

Lopukhine N., Crawhall N., Dudley N., Figgis P., Karibuhoye C., Laffoley D., Londono M., MacKinnon K., & Sandwith T. (2012). Protected Areas: providing solutions to 21st century challenges. S.a.P.I.E.N.S, 5, 117–131.

Lorimer, J. (2012). Multinatural geographies for the Anthropocene. Progress in Human Geography, 36, 593-612

Lowe, J. A., & Gregory, J. M., (2006). Understanding projections of sea level rise in a Hadley Centre coupled climate model, Journal of Geophysical. Research., 111, C11014.

Malhi, Y., Aragão, L. E., Galbraith, D., Huntingford, C., Fisher, R., ZElazowski, P., Sitch, S., McSweeney, C. & Meir, P. (2009). Exploring the likelihood and mechanism of a climate-change-induced dieback of the Amazon rainforest. Proceedings of the National Academy of Sciences, 106, 20610-20615.

Malkin, E., (2014). Lawlessness is undoing effort to save Honduran forests. The New York Times. Available from: http://www.nytimes.com/2014/02/13/world/americas/lawlessness-undoing-effort-to-save-honduran-forests.html?_r=0 [Accessed: 24 July 2015]

Mansourian, S., & Dudley, N. (2008). Public funds to protected areas. WWF International, Gland, Switzerland.

Marris, E. (2013) Rambunctious Gardens. Bloomsbury, UK.

Marris, E. (2013). Rambunctious garden: saving nature in a post-wild world, Bloomsbury Publishing USA.

Mascia M.B., Pailler S., Krithivasan R., Roshchanka V., Burns D., Mlotha M.J., Murray D.R., & Peng N. (2014) Protected area downgrading, downsizing, and degazettement (PADDD) in Africa, Asia, and Latin America and the Caribbean, 1900–2010. Biological Conservation, 169, 355–361.

Mascia, M. B., & Pailler, S. (2011). Protected Area downgrading, downsizing, and degazettement (PADDD) and its conservation implications. Conservation letters,4(1), 9-20

May, P.H., Gebara M.F., Conti, B.R., LimaG.R., (2013). The 'Ecological' Value Added Tax (ICMS-Ecológico) in Brazil and its effectiveness in State biodiversity conservation: a comparative analysis. Paper presented at 12th Biennial Conference of the International Society for Ecological Economics ISEE 2012: ECOLOGICAL ECONOMICS AND RIO +20. Panel session: The role of economic instruments in the conservation policy mix

Mázsa, K., Balázs, B., Bölöni, J., & Horváth, F. (2013). Strict Forest Reserve Research in the Margin of the Carpathians, the Vár-hegy Case-Study. In The Carpathians: Integrating Nature and Society Towards Sustainability (pp. 257-269). Springer Berlin Heidelberg.

McCarthy, J. (2000). The changing regime: Forest property and reformasi in Indonesia. Development and Change, 31, 91-129.

McCarthy, J. F. (2002). Turning in circles: district governance, illegal logging, and environmental decline in Sumatra, Indonesia. Society &Natural Resources, 15, 867-886.

McDonald, M., Smith, B. and Ward, K. (2006). Marketing due diligence: Reconnecting Strategy to share price, Butterworth-Heinemann.

McElhinny, V. (2007). Second Latin American Congress of National Parks and Protected Areas. San Carlos de Bariloche, Argentina, 1–4 Oct 2007. Bank Information Center, Info Brief, November 2007.

McLeod, Elizabeth and Salm, Rodney V. (2006). Managing Mangroves for Resilience to Climate Change. IUCN, Gland, Switzerland. 64pp.

McNeely, J. A., & Schutyser, F. (2003). Protected Areas in 2023: scenarios for an uncertain future. IUCN, Glad, Switzerland.

McNeely, J.A. (2005). Friends for Life: New partners in support of protected areas. IUCN, Gland, Switzerland and Cambridge, UK.

McSweeney, K., Nielsen, E. A., Taylor, M. J., Wrathall, D. J., Pearson, Z., Wang, O., & Plumb, S. T. (2014). Drug policy as conservation policy: Narco-deforestation. Science, 343 (6170), 489-490.

Medeiros, R. & Young; C.E.F. (2011). Contribuição das unidades de conservação brasileiras para a economianacional: Relatório Final. Brasília: UNEP WCMC, 120p.

Merrill M. (1990) Yellowstone and the Great West: Journals, Letters, and Images from the 1971 Hayden Ex pedition. University of Nebraska Press, USA.

Meunier, G., & Lavoie, C. (2012). Roads as corridors for invasive plant species: New evidence from Smooth Bedstraw (Galium mollugo). Invasive Plant Science and Management, 5(1), 92-100.

Meyers, J. & Walker, B. H. (2003). Thresholds and alternate states in ecological and social-ecological sys tems: thresholds database, Resilience Alliance, URL: http://www.resalliance.org.au. [Accessed: 25 August 2015]

Millennium Ecosystem Assessment (2005). MEA: Ecosystems and Human Well-being: synthesis, Island Press, Washington, DC

Mittermeier, R. A., Myers, N., Thomsen, J. B., DA Fonseca, G. A. & Olivieri, S. (1998). Biodiversity hotspots and major tropical wilderness areas: approaches to setting conservation priorities. Conservation biology, 12, 516-520.

MMA, (2015). Ministério do Meio Ambiente Website. Available at: http://www.mma.gov.br [Accessed: 7 August 2015]

Molnar, Augusta, Scherr, Sara J. and Khare, Arvind. (2004). Who Conserves the World's Forests? Community-Driven Strategies to Protect Forests & Respect Rights. Forest Trends: Washington, DC.

Moran, D. (1994). Contingent valuation and biodiversity: measuring the user surplus of Kenyan protected areas. Biodiversity & Conservation, 3(8), 663-684.

Moran, E.F., Adams, R., Bakoyéma, B. Adams, R., Stefano, F., Boucek, B. (2006). Human Strategies for Coping with El Niño Related Drought in Amazônia. Climatic Change, 77(3-4) 343-361.

Moutinho, P., Osvaldo, S., Lima, A., Christovam, M., Alencar, A., Castro, I. & Nepstad, D., (2011). REDD in Brazil: A focus on the Amazon. Principles, criteria, and institutional structures for a national program for Reducing Emissions from Deforestation and Forest Degradation – REDD. Brasília, DF: Center for Strategic Studies and Management.

Munday, P.L., Dixson, D.L., McCormich, M.I., Meekan, M., Ferrari, M.C.O., Chivers, D.P. (2010) Replenishment of fish populations is threatened by ocean acidification. Oceans,107(29), 12930–12934

Myers, N. (1998). Lifting the veil on perverse subsidies. Nature, 392, 327-328.

Myers, N., Mittermeier, R. A., Mittermeier, C. G., Da Fonseca, G. A. & Kent, J. (2000). Biodiversity hotspots for conservation priorities. Nature, 403, 853-858.

Naredo J.M. (2003). La economía en evolución: Historia y perspectivas de las categorías básicas del pensamiento económico. 612.

Nelson, F. and Agrawal, A. (2008). Patronage or Participation? Community-based Natural Resource Management Reform in Sub-Saharan Africa. Development and Change 39(4): 557–585

Nelson, F., R. Nshala, & W.A. Rodgers (2007). The evolution and reform of Tanzanian wildlife management. Conservation and Society, 5(3), 232-61.

Nepstad, D. C., Verssimo, A., Alencar, A., Nobre, C., Lima, E., Lefebvre, P., & Brooks, V. (1999). Large-scale impoverishment of Amazonian forests by logging and fire. Nature, 398(6727), 505-508.

Nepstad, D. C., Verssimo, A., Alencar, A., Nobre, C., Lima, E., Lefebvre, P., Schlesinger, P., Potter, C., Moutinho, P. & Mendoza, E. (1999). Large-scale impoverishment of Amazonian forests by logging and fire. Nature, 398, 505-508.

Neumann R.P. (1998) Imposing Wilderness: Struggles over livelihood and nature preservation in Africa. University of California Press: USA.

Nicholls R.J & Mimura, N., (1998). Regional issues raised by sea-level rise and their policy implications. Climate Research, 11, 5-18.

Nicholls, M. (2014). Climate Change: Implications for Tourism. Key Findings from the Intergovernmental Panel on Climate Change Fifth Assessment Report. Cambridge University Press, UK.

Nielsen, H., & Spenceley, A. (2011). The success of tourism in Rwanda: Gorillas and more.

Nogueira-Neto, P. & de Melo Carvalho, J.C. (1979) A programme of ecological stations for Brazil. Environmental Conservation, 6, 95-104.

Noss, A. J. (1997). Challenges to nature conservation with community development in central African forests. Oryx, 31, 180-188.

Noss, R.F. (2001). Beyond Kyoto: forest management in a time of rapid climate change. Conservation Biology, 15(3), 578-590

NPS. (2015) Visitor spending effects: Yellowstone National Park. Available at: http://www.nature.nps.gov/socialscience/nps-park.cfm?park=421. [Accessed: 27 August 2015].

NSF. (2015). National Science Foundation Awards search – Tanzania. Available at: http://www.nsf.gov/awardsearch/simpleSearchResult?queryText=tanzania. [Accessed: 3 September 2015]

Odum, HT. (1972). Use of energy diagrams for environmental impact statements. In Proceedings of the Conference Tools of Coastal Management, 197-231. Marine Technology Society. Washington, D.C., USA.

OECD (2003). Steps Taken and Planned Future Actions by Participating Countries to Ratify and Implement the Convention on Combating Bribery of Foreign Public Officials in International Business Transactions, Organisation for Economic Cooperation and Development.

OECO, (2015). O que é o ICMS Ecológico. Available at: http://www.oeco.org.br/dicionario-ambiental/28048-o-que-e-o-icms-ecologico/ [Accessed: 23 August 2015].

Oglethorpe, J., Shambaugh, J. and Kormos, R. (2004). Parks in the crossfire: strategies for effective conservation in areas of armed conflict. Parks. 14(1): 2–8.

Okapi Conservation Project. (2012). Okapi conservation project, Epulu Update—June 28, 2012." June 28. [Accessed: 1 October 2014]. www.okapiconservation.org/news/okapi-conservation-project-epulu-update-june-28-2012/

Oldfield, S. (2003) eds. Trade in Wildlife : Regulation for Conservation. Earthscan, UK.

Oldfield, S. (2013). The trade in wildlife: regulation for conservation, Routledge.

Ooi, M. K. J., Auld, T. D. And Denham, A. J. (2009). Climate change and bet-hedging: interactions between increased soil temperatures and seed bank persistence. Global Change Biology, 15, 2375–2386.

Ordway, E. M. (2015). Political shifts and changing forests: Effects of armed conflict on forest conservation in Rwanda. Global Ecology and Conservation, 3, 448-460.

Ortwin, R., & Regina, S. (2012). OECD Reviews of Risk Management Policies Social Unrest. OECD Publishing.

Pabon-Zamora, L., J. Bezaury, F. Leon, L. Gill, S. Stolton, A. Grover, S. Mitchell and N. Dudley. (2008). Nature's Value: Assessing protected area benefits. Quick Guide Series, ed. J. Ervin. Arlington, VA: The Nature Conservancy.

Pachauri, R.K., and Reisinger, A., (eds.) (2007). Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. In Climate Change 2007: Synthesis Report., Geneva, Switzerland: IPCC.

Palmer, C. (2001). The extent and causes of illegal logging: An analysis of a major cause of tropical deforestation in Indonesia.

Parker, C., Cranford, M., Oakes, N., Leggett, M. (eds.) (2012). The Little Biodiversity Finance Book. Global Canopy Programme, Oxford.

Parmesan, C. & Yohe, G. (2003). A globally coherent fingerprint of climate change impacts across natural systems. Nature, 421, 37-42.

Parry, M., Canziani, O., Palutikof, J., van der Linden, P., Hanson, C., (eds.) (2007) Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press.

Partow, H. (2001). The Mesopotamian Marshlands: Demise of an Ecosystem. Early Warning and Assessment Technical Report. Kenya: UNEP.

Perlman D.L. & Adelson G. (1997). Biodiversity: exploring values and priorities in conservation. John Wiley & Sons., UK.

Perrings, C. & Pearce, D. (1994). Threshold effects and incentives for the conservation of biodiversity. Environmental and Resource Economics, 4, 13-28.

Perrings, C. (2009) The economics of ecosystem services. In The Princeton Guide to Ecology (ed. by S.Levin), pp. 652-659. Princeton University Press, Princeton.

Perry, P. J. (1997). Political corruption and political geography. , Aldershot, UK: Ashgate.

Pfueller, S. L., Lee, D., & Laing, J. (2011). Tourism partnerships in protected areas: exploring contributions to sustainability. Environmental management,48(4), 734-749.

Phillips, L.D. & Costa, C.A.B (2005) Transparent prioritisation, budgeting and resource allocation with multi-criteria decision analysis and decision conferencing. Department of Operational Research London School of Economics and Political Science. Working Paper LSEOR 05.75. LSE, London.

Pickering, C., Hill, W., Byrne, J. (2015) The 'hollow-middle': why positive community perceptions do not translate into pro-conservation behaviour in El Vizcaíno Biosphere Reserve, Mexico. International Journal of Biodiversity Science, Ecosystem Services & Management, 11(2), 168 – 183.

Plumptre, A. J. & Reynolds, V. (1994). The effect of selective logging on the primate populations in the Budongo Forest Reserve, Uganda. Journal of Applied Ecology, 631-641.

Plumptre, A. J., Bizumuremyi, J. B., Uwimana, F., & Ndaruhebeye, J. D. (1997). The effects of the Rwandan civil war on poaching of ungulates in the Parc National des Volcans. Oryx, 31(4), 265-273.

Pokhrel, Y. N., Fan, Y. & Miguez-Macho, G. (2014). Potential hydrologic changes in the Amazon by the end of the 21st century and the groundwater buffer. Environmental Research Letters, 9, 084004.

Polyakov, I.V., Timokhov, L.A., Alexeev, V.A., Bacon, S., Dmitrenko, I.A., Fortier, L., Frolov, I.E., Gascard, J., Hansen, E., Ivanov, V.V., Laxon, S., Mauritzen, C., Perovich, D., Shimada, K., Simmons, H.L., Sokolov, V.T., Steele, M., Toole, J., (2010). Arctic Ocean warming contributes to reduced polar ice cap. Journal of Physical Oceanography, 40, 2743–2756.

Poulter, B., Hattermann, F., Hawkins, E., Zaehle, S., Sitch, S., Restrepo Coupe, N., Heyder, U. & Cramer, W. (2010). Robust dynamics of Amazon dieback to climate change with perturbed ecosystem model parameters. Global Change Biology, 16, 2476-2495.

Primack, R.B (2014). Essentials of Conservation Biology. Sixth Edition., Sinauer Associates, MA, USA.

Primack, R.B. (2010) Essentials of Conservation Biology, 5th ed.; Sinauer Associates, Inc.: Sunderland, MA, USA.

PR Newswire. (2015) Carnival Foundation's $2.5 Million Donation Helping The Nature Conservancy Make Major Strides in Preserving Health of World's Oceans and Seas. Available at: http://www.prnewswire.com/news-releases/carnival-foundations-25-million-donation-helping-the-nature-conservancy-make-major-strides-in-preserving-health-of-worlds-oceans-and-seas-300121294.html [Accessed: 7 November 2015]

Radford, J. Q. & Bennett, A. F. (2004). Thresholds in landscape parameters: occurrence of the white-browed treecreeper Climacteris affinis in Victoria, Australia. Biological Conservation, 117, pp. 375–391.

Rahel, F. J. And Olden, J. D. (2008), Assessing the effects of climate change on aquatic invasive species. Conservation Biology, 22: 521–533.

Raman, T. R. S. (2011). Framing ecologically sound policy on linear intrusions affecting wildlife habitats. Background paper for the National Board for Wildlife. Submitted to the Standing Committee of the National Board for Wildlife during its 21st meeting on 20 January 2011.

Rasker, R., and A. J. Hansen. (2000). Natural amenities and population growth in the Greater Yellowstone region. Human Ecology Review, 7, 30–40.

Red-monitor (2013). More corruption involving Norwegian REDD funding in Tanzania? Available at: http://www.redd-monitor.org/2013/02/06/more-corruption-involving-norwegian-redd-funding-in-tanzania/ [Accessed: 3 September 2015]

Reid, A., Teamey, K. & Dillon, J. (2002). Traditional ecological knowledge for learning with sustainability in mind. Trumpeter, 18.

Reilly, J. et al. (1996). Agriculture in a changing climate. In Watson, Climate Change 1995: Impacts.

Reiter, P. (2001). Climate change and mosquito-borne disease. Environmental Health Perspective, 109 (1),141–161.

Ribeiro, D. (2000). The Brazilian People: The Formation and Meaning of Brazil. University Press of Florida, USA.

Rice, T. & Counsell, S. (1998). European Commission development funding: environmental and social impacts in tropical forest areas - an investigation by the Rainforest Foundation. London, Rainforest Foundation.

Ritchie, E. G. et al. (2013) Continental-scale governance and the hastening of loss of Australia's biodiversity. Conservation Biology, 27, 1133–1135.

Roberts, C. M., McClean, C. J., Veron, J. E., Hawkins, J. P., Allen, G. R., McAllister, D. E., Mittermeier, C. G., Schueler, F. W., Spalding, M. & Wells, F. (2002). Marine biodiversity hotspots and conservation priorities for tropical reefs. Science, 295, 1280-1284.

Robertson, J. & Van Schaik, C. P. (2001). Causal factors underlying the dramatic decline of the Sumatran orang utan. Oryx, 35, 26-38.

Robledano, F., Esteve, M.A., Farinos, P., Carreno, M.F., Martinez-Fernandez, J. (2010) Terrestrial birds as indicators of agricultural-induced changes and associated loss in conservation value of Mediterranean wetlands. Ecological Indicators, 10 (2), 274-286

Romero C., Athayde S., Collomb J.-G.E., DiGiano M., Schmink M., Schramski S., & Seales L. (2012) Conservation and Development in Latin America and Southern Africa: Setting the Stage. Ecology and Society, 17.

Rosen, G.E. and Smith, K.F. (2010). Summarizing the evidence on the international trade in illegal wildlife. EcoHealth, 7(1), 24-32

Rustad, S.A. and Binningsbø, H.M. (2012). A price worth fighting for? Natural resources and conflict recurrence. Journal of Peace Research 49:531–546.

Sachedina, H.T. (2008). Wildlife is our oil : conservation, livelihoods and NGOs in the Tarangire ecosystem , Tanzania. Word Journal Of The International Linguistic Association. University of Oxford.

Safina, C. & Klinger, D. H. (2008). Collapse of bluefin tuna in the western Atlantic. Conservation Biology, 22, 243-246.

Sampayo., E.M., Ridgway, T., Bongaerts, P., & Hoegh-Guldberg, O. (2008). Bleaching susceptibility and morality of corals are determined by fine-scale differences in symbiont type. PNAS. 105(30) pp. 10444-9

Santos J., Sato M., Pires J., & Maroti P. (2000) Environmental education in practice: The case of the ecological station of Jatai, Luiz Antonio, State of Sao Paulo, Brazil. Gestion Ambiental.

Sarre, S., Smith, G. T. & Meyers, J. A. (1995). Persistence of two species of gecko (Oedura reticulata and Gehyra variegata) in remnant habitat. Biological Conservation, 71, 25-33.

Scheffer, M., Hosper, S. H., Meijer, M. L., Moss, B. & Jeppesen, E. (1993). Alternative equilibria in shallow lakes. Trends in Ecology & Evolution, 8, 275-279.

Schubert, R., & Schellnhuber, H. J. (2009). Climate Change as a Security Risk. Routledge.

Schultz L, Duit A, Folke C. (2011). Participation, adaptive co-management, and management performance in the world network of biosphere reserves. World Development, 39, 662– 671.Scotland, N., Smith, J., Hikma, L., Hiller, M., Jarvis, B., Kaiser, C., Leighton, M., Paulson, L., Pollard, E., Ratnasari, D., Ravanell, R., Stanley, S., Erwidodo, Currey, D. A. & Setyarso, A. (2000). Indonesia Country Paper on illegal logging. Prepared for the World Bank-World Wide Fund for Nature Workshop on Control of Illegal Logging in East Asia. 28th August 2000., Jakarta, Indonesia. .

Scott R. L, Philip B. D., Healy H., Gregory P. A., Christopher B. F. & David D. A. (2009). The velocity of climate change. Nature, 462, 1052-1055.

Secretariat of the Convention on Biological Diversity (2008). Protected areas in todays' world: Their values and benefits for the welfare of the planet. Diversity, 43, 96.

Sendalo, D.S.C. (2009). A review of land tenure policy implications on pastoralism in Tanzania. Department of Livestock Research, Training and Extension, Ministry of Livestock Development and Fisheries, Dares Salam, Tanzania

Siegert, F., Ruecker, G., Hinrichs, A., & Hoffmann, A.A. (2001). Increased damage from fires in logged forests during droughts caused by El Niño. Nature, 414, 437-440.

Simpson, S.D., Munday, P.L., Wittenrich, M.L., Manassa, R., Dixson, D.L., Gagliano, M., Yan, H.Y. (2011). Ocean acidification erodes crucial auditory behaviour in a marine fish. Biologiy Letters, l20110293

Smith, J., Obidzinski, K., Subarudi, S. & Suramenggala, I. (2003). Illegal logging, collusive corruption and fragmented governments in Kalimantan, Indonesia. International Forestry Review, 5, 293-302.

Smith, R. J. & Walpole, M. J. (2005). Should conservationists pay more attention to corruption? Oryx, 39, 251–256.

Smith, R. J., Muir, R. D., Walpole, M. J., Balmford, A. & Leader-Williams, N. (2003). Governance and the loss of biodiversity. Nature, 426, 67-70.

Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, R.B., Tignor, M., Miller, H.L. (eds.) (2007). Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge: Cambridge University Press.

Spash, C.L. and Aslaksen, L. (2014) Re-establishing an ecological discource in the policy debate over how to value ecosystems and biodiversity. SRE – Discussion Papers, 2014/05. WU Vienna University of Economics and Business, Vienna.

Speight, M.R., & Henderson, P., (2010). Marine ecology: concepts and applications. John Wiley & Sons.

SPUR. (2009). Strategies for Managing Sea Level Rise. The Urbanist, 487.

Standing, A. (2008). Corruption and industrial fishing in Africa. U4 Issue, 2008.

Stapenhurst, R. & Kpundeh, S. J. (1999). Curbing corruption: Toward a model for building national integrity, World Bank Publications.

Stellmacher, T., Winter, E., Grote, U. (2012) Protected areas in Tanzania: from ineffective management to PA certificates? Proceedings of the Conference on the Human Dimensions of Global Environmental Change. Berlin, 5-6 October 2012. Institute for Environmental Economics and World Trade, Leibniz University, Hannover

Stockholm Resilience Centre. (2015). Available: http://www.stockholmresilience.org/21/about.html (Accessed: 10 September 2015).

Stolton S. & Dudley N. (2010) Arguments for Protected Areas: Multiple Benefits for Conservation and Use. Routledge, USA.

Sullivan, S. (2013). Banking nature? The spectacular financialisation of environmental conservation. Antipode, 45(1), 198-217.

T'Sas-Rolfes, M, J. (1995). Rhinos: Conservation, Economics and Trade-Offs, Institute of Economic Affairs, London

TEEB (2008). The economics of ecosystems and biodiversity – an interim report. European Commission, Brussels.

The Institute of Foresters of Australia. (2006). Managing fire in Australian forests and woodlands. Forest Policy Statement No. 3.2

Theisen, O. M., Gleditsch, N. P., & Buhaug, H. (2013). Is climate change a driver of armed conflict?. Climatic Change, 117(3), 613-625.

Thomas, C.D., Cameron, A., Green, R.E., Bakkenes, M., Beaumont, L.J., Collingham, Y.C., Erasmus, B.F.N, de Siqueira, M.F., Grainger, A., Hannah, L., Hughes, L., Huntley, B., van Jaarsveld, A.S., Midgley, G.F., Miles, L., Ortega-Huerta, M.A., Phillips, O.L., & Williams, S.E. (2004). Extinction risk from climate change. Nature, 427, 145-148.

Thompson, M. & Homewood, K. (2002). Entrepreneurs, elites, and exclusion in Maasailand: trends in wildlife conservation and pastoralist development. Human Ecology, 30, 107-138.

TRAFFIC, (2008). http://www.traffic.org/trade/ [Accessed: 26 September 2014].

UNEP, (1994). United Nations Environment Programme. Assessment and monitoring of climatic change impacts on mangrove ecosystems. UNEP Regional Seas Reports and Studies. Report no. 154.

UNESCO. (2013). The director-general of UNESCO expresses deep concern at the rising violence in the Central African Republic, at the heart of the Sangha Trinational Site. http://whc.un esco.org/en/news/1006. [Accessed: 2 September 2014]

United Nations (1992) United Nations Framework Convention on Climate Change, New York.

USFWS (2007). Endangered and Threatened Wildlife and Plants; Notice of availability for draft recovery crediting guidance. US Fish and Wildlife Service, Department of the Interior. Federal Register, 72 (212)/ 62258-62264. http://www.fws.gov/endangered/esa-library/pdf/recovery%20crediting.pdf

Van der Werf, G.R., James T. Randerson, G. James Collatz, Louis Giglio, Prasad S. Kasibhatla, Avelino F. Arellano Jr., Seth C. Olsen, and Eric S. Kasischke. (2004). Continental-Scale Partitioning of Fire Emissions During the 1997 to 2001 El Niño/La Niña Period. Science, 303(5654), 73-76.

Van der Ree, R., Bennett, A. F. & Gilmore, D. C. (2004). Gap-crossing by gliding marsupials: thresholds for use of isolated woodland patches in an agricultural landscape. Biological Conservation, 115, 241–249.

VCA. (2015). Conserving our planet, hectare by hectare. Available at: http://www.v-c-a.org/. [Accessed: 28 January 2015].

Veit, P. G., & Benson, C. (2004). When parks and people collide. Human Rights Dialogue: Environmental Rights.

Velho, N., Agarwala, M., Srinivasan, U., & Laurance, W. F. (2014). Collateral damage: impacts of ethno-civil strife on biodiversity and natural resource use near Indian nature reserves. Biodiversity and conservation, 23(10), 2515-2527.

Vernon R. B., (2010).The contribution of hunting tourism: How significant is this to national economies? in Contribution of Wildlife to National Economies. Joint publication of FAO and CIC. Budapest. 72 pp.

Viner, D. & Agnew, M. (1999). Climate Change and Its Impacts on Tourism. Report Prepared for WWF-UK, Climatic Research Unit.

Vreugdenhil, D., Terborgh, J., Cleef, A.M., Sinitsyn, M., Boere, G.D.,Archaga, V.L., Prins, H.H.T. (2003). Comprehensive Protected Areas System Composition and Monitoring. Shepherdstown (WV): World Institute for Conservation and Environment.

Waldron, A, Mooers A.O., Miller, D.C., Nibbelink, N., Redding, D., Kuhn, T.S., Roberts, J.T., Gittleman, J.L., (2013), Targeting global conservation funding to limit immediate biodiversity declines. Proceedings of the National Academy of Sciences of the United States of America, 110(29), 12144-12148.

Walker, B. & Salt, D. (2006). Resilience Thinking: Sustaining ecosystems and people in a changing world. Island Press, Washington

Walker, W., Baccini, A., Schwartzman, S., Ríos, S., Oliveira-MIRANDA, M. A., Augusto, C., Ruiz, M. R., Arrasco, C. S., Ricardo, B. & Smith, R. (2015). Forest carbon in Amazonia: the unrecognized contribution of indigenous territories and protected natural areas. Carbon Management, 1-7.

Walpole, M. (2003). Wildlife and people: conflict and conservation in Masai Mara, Kenya. IIED, London, UK.

Walpole, M., & Leader-Williams, N. (2002). Tourism and flagship species in conservation. Biodiversity and conservation, 11(3), 543-547.

Walther, G.R. Eric, P., Peter, C., Menzel, A., Camille, P., Trevor J. C. Beebee, Fromentin, J., Hoegh-Guldberg,). & Bairlein, F. (2002). Ecological responses to recent climate change. Nature, 416, 389-395.

Wardle, D. A., Bardgett, R. D., Callaway, R. M. & Van der Putten, W. H. (2011). Terrestrial ecosystem responses to species gains and losses. Science, 332, 1273–1277.

Warne, K. (2014). 5 Takeaways From Huge Once-a-Decade Gathering on World's Protected Areas. [ONLINE] Available at: http://news.nationalgeographic.com/news/2014/11/141121-world-parks-congress-conservation-nature-science/. [Accessed: 01 September 2015].

Watkins, H., Hawkins, K. & Cormack, A. (2014) HS2: The Case for a Greener Vision. The Wildlife Trusts, UK.

Watson J.E.M., Dudley N., Segan D.B., & Hockings M. (2014). The performance and potential of protected areas. Nature, 515, 67–73.

WCS, (2013). Wildlife Conservation Society. Tigers in peril. Available at: www.wcs.org/tigers/ [Accessed: 10 September 2015].

Westman, W.E. (1977). How much are nature's services worth? Science, 197, 960–964.

Willis, K. J., Bailey, R. M., Bhagwat, S. A. & Birks, H. J. B. (2010). Biodiversity baselines, thresholds and resilience: testing predictions and assumptions using palaeoecological data. Trends in Ecology & Evolution, 25, 583-591.

Wissel, C. (1984). A universal law of the characteristic return time near thresholds. Oecologia, 65, pp. 101–107.

Wittemyer, G., Elsen, P., Bean, W. T., Burton, A. C. O., & Brashares, J. S. (2008). Accelerated human population growth at protected area edges.Science, 321(5885), 123-126.

WLT. (2012) Tropical forests and wetlands, Guatemala. World Land Trust. Available at: http://www.worldlandtrust.org/projects/guatemala [Accessed: 8 November 2015]

Woodley, S., Bertzky, B., Crawhall, N., Dudley, N., Londoño, J. M., MacKinnon, K., & Sandwith, T. (2012). Meeting Aichi Target 11: what does success look like for protected area systems?. Parks, 18(1), 23-36.

World Bank (2015) http://www.worldbank.org/en/country/tanzania/projects/operational-documents?qterm=&teratopic_exact=Water+Resources&docty_exact=Project+Appraisal+Document [Accessed: 7 September 2015]

World Bank. (2005). Pilot program to conserve the Brazilian rain forest (PPG7). Results profile. Washington, DC : World Bank Group. Available at: http://documents.worldbank.org/curated/en/2005/12/23966829/pilot-program-conserve-brazilian-rain-forest-ppg7 [Accessed: 23 August 2015]

World Economic Forum (2015) Global Risks 2015, Geneva: World Economic Forum.

World Heritage, (2014). Environmental NGOs back up IUCN position on no-go for extractives in World Heritage sites. Available at: http://worldparkscongress.org/drupal/node/221 [Accessed: 3 November 2015]

World Travel & Tourism Council (2015). Travel & Tourism Economic Impact 2015 Tanzania. Harlequin Building, London.

Wright, S. J, Sanchez-Azofeifa, A, Portillo-Quintero, C and Davies, D (2007). Poverty and corruption compromise in Tropical Forest Reserves. Ecological Applications, 17: 1259–1266.WWF & TCME (2012) Integrating Environment into Investment Decisions: Introductory Guidance for Tanzania's Mining Sector.

WWF (2006). Free-flowing Rivers: Economic Luxury or Ecological Necessity? WWF, Zeist, Netherlands.

WWF METT & GD-PAME (2015). Management Effectiveness Tracking Tool, and Global Database on Protected Area Management Effectiveness. Databases provided by UNEP-WCMC.

WWF, (2006). A future for protected areas in the Brazilian Amazon Available at: http://wwf.panda.org/what_ we_do/where_we_work/amazon/vision_amazon/models/amazon_protected_areas/financing/arpa/ [Accessed: 23 August 2015]

WWF, (2014). Brazilian System of Protected Areas (SNUC) completes 14 years of advances and challenges. Available at: http://www.wwf.org.br/informacoes/english/?40662/Brazilian-System-of-Protected-Areas-SNUC-completes-14-years-of-advances-and-challenges [Accessed: 7 August 2015].

WWF. (2014). http://www.wwf.org.uk/what_we_do/illegal_wildlife_trade/[Accessed: 2 September, 2014]

WWF. (2015). PADDDtracker: Tracking Protected Area Downgrading, Downsizing, and Degazettement. Available at: http://www.padddtracker.org/ [Accessed: 18 September 2015]

Xu, J., & Melick, D. R. (2007). Rethinking the effectiveness of public protected areas in southwestern China. Conservation Biology, 21(2), 318-328.

Y. J. Kaufman, Y.J. C. J. Tucker, and I. Y. Fung, (1989). Remote sensing of biomass burning in the tropics. Advances in Space Research, 9, 265–268.

Zimmerman, M.E. (2003). The black market for wildlife: Combating transnational organized crime. Vanderbilt Journal of Transnational Law, 36 (5), 1154-1689.

STRANDED
ASSETS

PROGRAMME

Smith School of Enterprise and the Environment
University of Oxford
South Parks Road
Oxford, OX1 3QY
United Kingdom

E enquiries@smithschool.ox.ac.uk
T +44 (0)1865 614963
F +44 (0)1865 275885
www.smithschool.ox.ac.uk/research/stranded-assets/