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Can biodiversity offsetting follow in carbon's footprints?

by

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Abstract

1. Biodiversity is the natural capital on which global economies are built. Despite this, biodiversity is still in decline, drawing ever more frequent comparisons to Earth's five mass extinction events. Offsetting residual biodiversity impacts of damaging projects has been proposed by many to combat one of the main drivers of this decline as identified by the Millennium Ecosystem Assessment: land use change as a result of economic growth and development. This literature review assesses the potential for a global biodiversity offsets market by comparison with the global carbon market, arguably the most successful environmental market in terms of market value and volumes traded.
2. The current state of both carbon and biodiversity offset markets is reviewed, and their relative strengths and weaknesses analysed. Possible factors contributing to the success of the carbon market are identified; these are then mapped onto the biodiversity market in an attempt to predict whether it can follow the same trajectory as carbon offsetting, especially with regards to the establishment of a biodiversity disclosure project.
3. The global carbon market, as driven by the European Union Emissions Trading Scheme (EU ETS), has matured into a significant environmental market, with a market value of US$176 billion and 10,281 $MtCO_2$ volume traded. The market, especially the voluntary component, has been heavily criticised for lack of quality assurance. The factors contributing to carbon's success were identified to be the homogeneity of emissions, high volume of carbon research, quantitative nature, timing, and offset location. The global biodiversity market is very much smaller, with a market value of US$2.4-$4 billion and 187,000 hectares protected or restored. Four biodiversity market case studies were analysed. The literature highlighted significant compliance and monitoring issues in the U.S.; this non-compliance has led to the failure to achieve the policy of no net loss both of wetland and wetland function.
4. This review found that the factors identified proved not to map well onto a potential biodiversity market. Biodiversity suffers from significant ignorance and sizeable gaps in knowledge, which does not align well with the quantitative nature of financial markets. Although a biodiversity market would share similar considerable institutional frameworks and environmental policy acceptance as the current carbon market, understanding of the complexity of biodiversity at the most basic level is found likely to prove a significant hindrance to the establishment of a global market. This review also found fertile ground for the creation of a biodiversity disclosure project, but no significant steps to seed one. The overwhelming conclusion is that biodiversity offsetting is unlikely to form a sizeable market comparable to carbon, and that biodiversity conservation efforts should continue to be focussed elsewhere.

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Abbreviations

A/R	Afforestation and reforestation
BBOP	Business & Biodiversity Offsets Partnership
BDP	Biodiversity disclosure project
BIP	Biodiversity Indicators Partnership
CA DFG	California Department of Fish and Game
CBD	Convention on Biological Diversity
CDM	Clean Development Mechanism
CDP	Carbon Disclosure Project
CDP WDP	Carbon Disclosure Project Water Disclosure Project
CER	Certified Emissions Reduction
CMP	Conservation Measures Partnership
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
COP17	17th Conference of the Parties
CRC	Carbon Reduction Commitment
CSE	Centre for Science and Environment
CSR	Corporate Social Responsibility
DEFRA	Department for Environment, Food and Rural Affairs
DNA	Deoxyribonucleic acid
EBRD	European Bank for Reconstruction and Development
EC	European Commission
eftec	Economics for the Environmental Consultancy
EU ETS	European Union Emissions Trading Scheme
EIA	Environmental Impact Assessment
EIB	European Investment Bank
ELI	Environmental Law Institute
EPL	Environmental Profit & Loss
ERU	Emission Reduction Unit
ESA	Endangered Species Act
EU	European Union
FCS	Favourable Conservation Status
GDP	Gross Domestic Product
GHG	Greenhouse gas
HGM	Hydrogeomorphic
IEEP	Institute for European Environmental Policy
ILF	In-lieu fee
IPBES	Intergovernmental Platform on Biodiversity & Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
LULUCF	Land use, land use change, and forestry
NGO	Non-Governmental Organisation
NMFS	National Marine Fisheries Service
NRC	National Research Council
NSW	New South Wales

NSW DECCW	New South Wales Department for Environment, Climate Change, and Water
NSW OEH	New South Wales Office of Environment and Heritage
MA	Millennium Ecosystem Assessment
OTC	Over-the-counter
PR	Public relations
REDD	Reducing Emissions from Deforestation and Forest Degradation
RIBITS	Regulatory and In-lieu fee and Bank Information Tracking System
RSPB	Royal Society for the Protection of Birds
SAC	Special Area of Conservation
SEA	Strategic Environmental Assessment
SER	Social and environmental reporting
SPA	Special Protection Area
TEEB	The Economics of Ecosystems and Biodiversity
UK	United Kingdom
UNESCO	United Nations Educational, Scientific and Cultural Organisation
UNEP-WCMC	United Nations Environment Programme-Wildlife Conservation Monitoring Centre
UNFCCC	United Nations Framework Convention on Climate Change
U.S.	United States
USACE	U.S. Army Corps of Engineers
USDOI	U.S. Department of the Interior
USFWS	U.S. Fish and Wildlife Service
WTA	Willingness to accept
WTP	Willingness to pay

CHAPTER ONE

1 INTRODUCTION

1.1 What is biodiversity?

Biological diversity (usually shortened to biodiversity) is defined in the CBD as:

"the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (CBD, 1992)

The most important aspect of this definition is the idea of components that contribute towards total biodiversity – a single species can contribute through the very existence of its species, its inherent genetic diversity, or its contribution to the functioning of its surrounding ecosystem.

Biodiversity represents our priceless natural inheritance; an inheritance that has been accumulating in the building society of planet earth for over 3.5 billion years. It is the platform on which all human economies are based.

1.2 Why is biodiversity important?

1.2.1 The moral imperative

Ironically, the most seemingly obvious argument for the importance of biodiversity to conservationists is the weakest with other stakeholders, with little workable clout: conservation as a moral imperative. Moral rationale for conservation veers dangerously close to the arrogance that the human race is the pinnacle of life on earth, but no one can deny we are undoubtedly one of the most populous and destructive species. However, it remains a crucial weapon in the conservationist's arsenal as tropical forests, and indeed other ecosystems, are more often than not "worth more dead than alive" (Terborgh, 1999, p. 18).

Besides accusations of green imperialism, the moral argument component of the protectionist paradigm has been criticized for masking differences in local and nonlocal perceptions, and the independent weights these carry (Wilshusen, et al., 2002).

Even if the moral argument takes hold, as discussed in IUCN and Shell's joint report *Building Biodiversity Business*, "Modern economies are very good at producing what people will pay for. They are not so good at preserving what is priceless" (Bishop, et al., 2008, p. 8). So when looking at the importance of biodiversity, it is crucial to look at what highly biodiverse regions can offer in terms of economic value.

1.2.2 Bioprospecting
One of the frontrunners in adding tangible value to biodiversity is the idea of bioprospecting. Bioprospecting can be defined as "the systematic search for new commercial applications for biota" (Barrett & Lybbert, 2006, p. 293). Thus the total value of the ecosystem can be theoretically assigned the value of all future products that could be discovered. Real valuations using this method tend to be based on new market agreements and contracts for bioprospecting with pharmaceutical companies (Christie, et al., 2006). A recent example of successful bioprospecting is the discovery that a gel derived from *Acmella oleracea*, a plant found in central Peru, could replace anaesthetic injections for dental work (Collins, 2012).

Bioprospecting suffers from 'Grimsby syndrome' in the respect that its name is the root of a lot of the criticism levelled against it. Bioprospecting, or biodiversity research as it is also known in an attempt to address this problem, is far from the directionless gold rushes of prospecting in the American West. Prospecting in the West was almost entirely based on luck and known for being ruthlessly unfulfilling despite the enormous effort needed. Bioprospecting, on the other hand, is still very much scientifically based; research is channelled towards avenues where the probability of discovery is expected to be high, whilst new technologies allow the cost of searching to be drastically reduced (Rausser & Small, 2000).

However, many scientists are entirely sceptical of bioprospecting and its use for international conservation: "With the possible exception of a few extraordinary sites, there is no hard empirical or theoretical evidence that bioprospecting adds significant value to tropical ecosystems" (Barrett & Lybbert, 2006, p. 295). Furthermore, Kursar points out that it also falls short on equity grounds, as benefits from commercialised products in the developing world are very rarely shared with the original source country (Kursar, 2011).

1.2.3 Resilience

Ecosystem (or ecological) resilience, first discussed by Canadian ecologist C. S. Holling (Holling, 1973), can be defined as "the capacity of an ecosystem to tolerate disturbance without collapsing into a qualitatively different state that is controlled by a different set of processes" (The Resilience Alliance, 2002). Ecological resilience is becoming an increasingly important aspect in order to safeguard against catastrophic changes associated with climate change (Bellard, et al., 2012).

Whilst the biodiversity of a system is only one factor that contributes towards ecological resilience, there has been a lot of research into whether higher species richness (one measure of biodiversity) increases the stability of an ecosystem (Peterson, et al., 1998). At the most basic level, it follows from simple evolutionary theory that more highly biodiverse ecosystems will have a higher ecological resilience: they will inevitably have a larger gene pool from which to select adaptive genotypes. Indeed, Darwin recognised this very early on when he suggested that a system is more ecologically stable with a large number of species (Darwin, 1859). The ecological resilience of an ecosystem is obviously dependent on species with similar functional roles responding differently to external pressures, and it follows that this is more likely with more species (Hughes, et al., 2005); there are a host of other studies linking high biodiversity with high resilience (Steneck, et al., 2002; Richards, et al., 2008; Willis, et al., 2010).

Evidence shows that this important function of biodiversity is lost and regime shifts to less productive states become more likely when biodiversity is reduced by anthropogenic activities (Folke, et al., 2004), e.g. removing sea otters from the kelp ecosystems off the NW coast of the United States (Steneck, et al., 2002).

1.2.4 Ecosystem services

One meta-analysis of the now substantial number of papers published concerning biodiversity and ecosystem functioning concludes that the "experimental evidence for a relationship between biodiversity and ecosystem process rates is compelling" (Balvanera, et al., 2006, p. 1). Another concludes with high confidence that: "Certain combinations of species are complementary in their patterns of resource use and can increase average rates of productivity and nutrient retention" (Hooper, et al., 2006). The dependence of the human race on the provision of ecosystem services is already well established (MA, 2005).

In 1997, Costanza and colleagues ambitiously attempted to place a value on the world's ecosystem services and natural capital (Costanza, et al., 1997). Their value, roughly US$33tr p.a., seems an enormous figure (global GDP was US$18tr); but as various criticisms have pointed out it is actually a vast *under*estimate. In terms of WTA (the paper breaks the defining assumption of WTP), global ecosystem services *have* to be assigned an infinite value, and as Toman points out, US$33tr p.a. is "a serious underestimate of infinity" (Toman, 1998, p. 58). Extending this, if biodiversity begets ecosystem functioning, then biodiversity itself is priceless. However, as aforementioned, a priceless valuation is not useful and ironically does more damage than good (Bishop, et al., 2008).

1.3 Threats

Global biodiversity faces a host of threats. Various authors have attempted to categorise these threats, including Jared Diamond's 'evil quartet' (Diamond, 1995) and E. O. Wilson's 'HIPPO' acronym (Wilson, 2002). The most comprehensive list of global threats (and their potential solutions) was released by the IUCN-CMP in 2008, including a total of eleven major threat categories (Salafsky, et al., 2008) (Table 1).

Table 1: IUCN-CMP classification of direct threats to biodiversity, adapted from (Salafsky, et al., 2008)

Threats by level of classification	Definition
1. Residential and commercial development	human settlements or other non-agricultural land uses with a substantial footprint
2. Agriculture and aquaculture	threats from farming and ranching as a result of agricultural expansion and intensification, including silviculture, mariculture, and aquaculture
3. Energy production and mining	threats from production of non-biological resources
4. Transportation and service corridors	threats from long, narrow transport corridors and the vehicles that use them including associated wildlife mortality
5. Biological resource use	threats from consumptive use of "wild" biological resources including deliberate and unintentional harvesting effects; also persecution or control of specific species
6. Human intrusions and disturbance	threats from human activities that alter, destroy and disturb habitats and species associated with non-consumptive uses of biological resources
7. Natural system modifications	threats from actions that convert or degrade habitat in service of "managing" natural or semi-natural systems, often to improve human welfare
8. Invasive and other problematic species and genes	threats from non-native and native plants, animals, pathogens/microbes, or genetic materials that have or are predicted to have harmful effects on biodiversity following their introduction, spread and/or increase in abundance
9. Pollution	threats from introduction of exotic and/or excess materials or energy from point and non-point sources
10. Geological events	threats from catastrophic geological events
11. Climate change and severe weather	long-term climatic changes that may be linked to global warming and other severe climatic or weather events outside the natural range of variation that could wipe out a vulnerable species or habitat

Another important resource is Sutherland and colleagues' annual horizon scan of potential direct threats to biodiversity (Sutherland, et al., 2012). Sutherland and colleagues' highlight potential emerging threats like synthetic meat and nanosilver in wastewater, but there is one threat that is omnipresent : habitat change and fragmentation (taking the form of 'Large-scale international land acquisitions' in the

2012 horizon scan, as well as being present in the IUCN-CMP's, Diamond's, and Wilson's threats). Lewis has even argued that predictions of the threat of climate change to biodiversity would pull already stretched conservation resources away from the more important objective of slowing and mitigating habitat destruction and degradation (Lewis, 2006).

1.4 Why does it need protection?

"We are losing biodiversity at an alarming rate" is now an ubiquitous phrase in biodiversity research. Current and projected future extinction rates are far above the long-term average (Figure 1). Many authors are already comparing the current rates of biodiversity loss as Earth's sixth mass extinction, or the 'Holocene mass extinction' (Thomas, et al., 2004; Ceballos & Ehrlich, 2002; Wake & Vredenburg, 2008; Barnosky, et al., 2011).

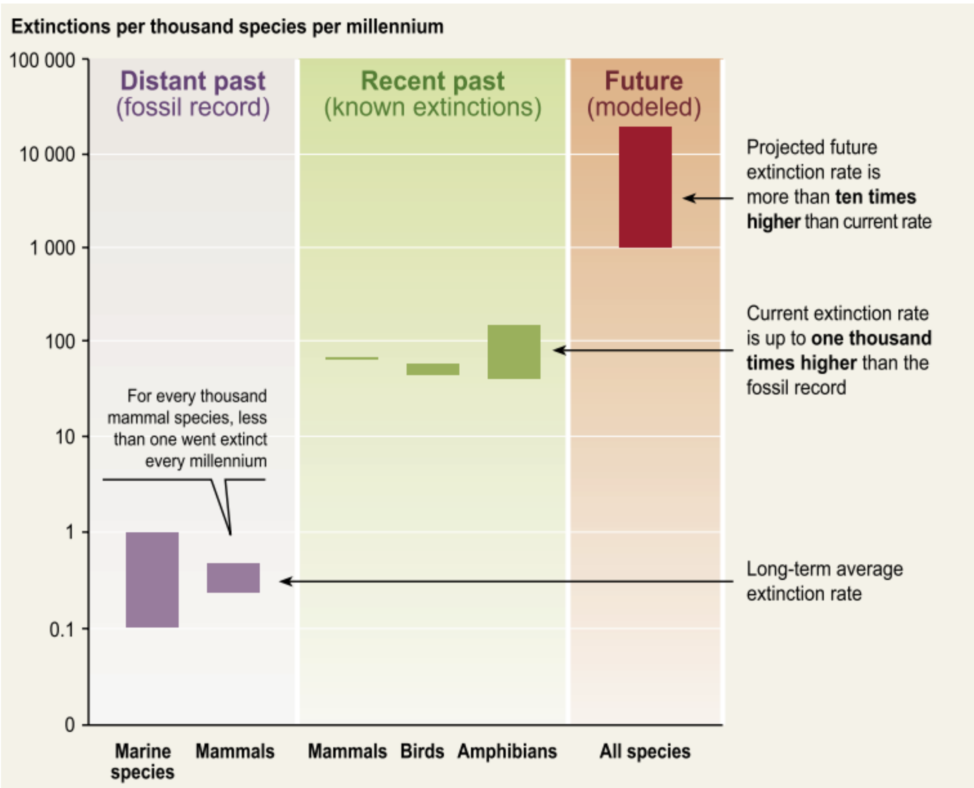

Figure 1: Extinctions per thousand species per millennium; Source: (MA, 2005)

Concerns regarding the loss of biodiversity stem largely from MacArthur and Wilson's theory of island biogeography (MacArthur & Wilson, 1967), and Tilman's development of the idea into the concept of the extinction debt (Tilman, et al., 1994). The extinction debt treats conservation reserves as 'islands' and describes the time lag between habitat perturbations and species extinctions (Figure 2). Recent worrying evidence for the theory has been found by Wearn and colleagues in the Brazilian Amazon:

"local extinctions of forest-dependent vertebrate species have thus far been minimal (1% of species by 2008), with more than 80% of extinctions expected to be incurred from historical habitat loss still to come" (Wearn, et al., 2012, p. 1).

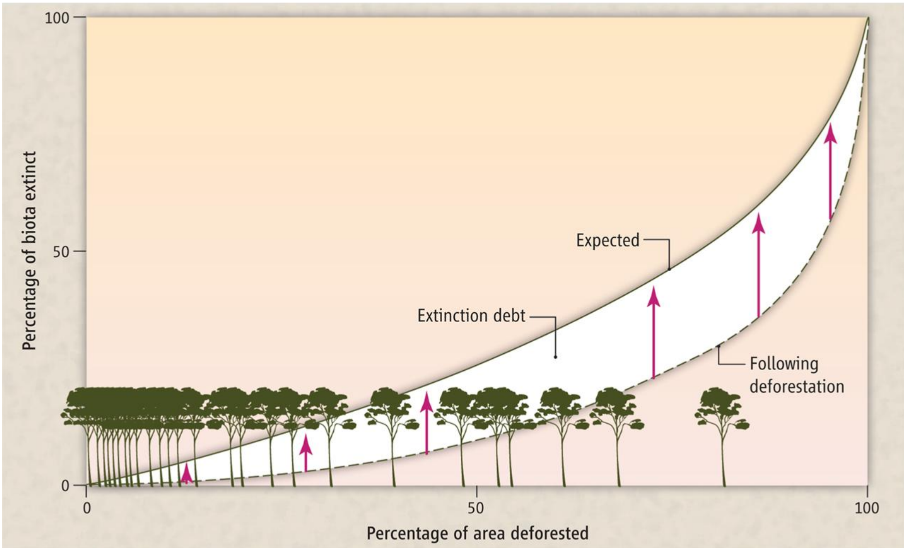

Figure 2: Deforestation extinction debt; Source: (Rangel, 2012)

Furthermore, it has been long known that ecosystems can exist in alternate stable states (Scheffer, et al., 2001); whether this be eutrophication of a small pond or the large scale change of a coral reef to an algal zone, even small scale perturbations of biodiverse systems can have catastrophic concomitant effects on the biodiversity of a system.

It is clear that biodiversity is incredibly important, and efforts need to be stepped up to protect it. One of the many options put forward is the concept of biodiversity offsetting (Parker, et al., 2012).

1.5 Biodiversity offsetting

The BBOP comprehensively defines biodiversity offsets as:

"measurable conservation outcomes of actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after

appropriate prevention and mitigation measures have been taken. The goal of biodiversity offsets is to achieve no net loss and preferably a net gain of biodiversity on the ground with respect to species composition, habitat structure, ecosystem function and people's use and cultural values associated with biodiversity" (BBOP, 2012)

Biodiversity offsetting can be summed up by three words: no net loss. No net loss developed from the United States' no net loss wetlands policy first adopted by the Bush Sr. administration after the introduction of the 1972 Clean Water Act. The policy was in response to the worrying reduction of U.S. wetlands from 215 million acres[3] in the 18th century to 99 million by the mid-1970s (Wilen & Frayer, 1990).

Crucial to the correct functioning of biodiversity offsetting is the concept of the mitigation hierarchy (Figure 3). In order for biodiversity offsetting not to become a "licence to destroy, [...] offsets must only be used to compensate for genuinely unavoidable damage" (Lawton, et al., 2010, p. 86). For an impact to be genuinely unavoidable, options must be explored to first avoid, minimise or rehabilitate/restore (Table 2).

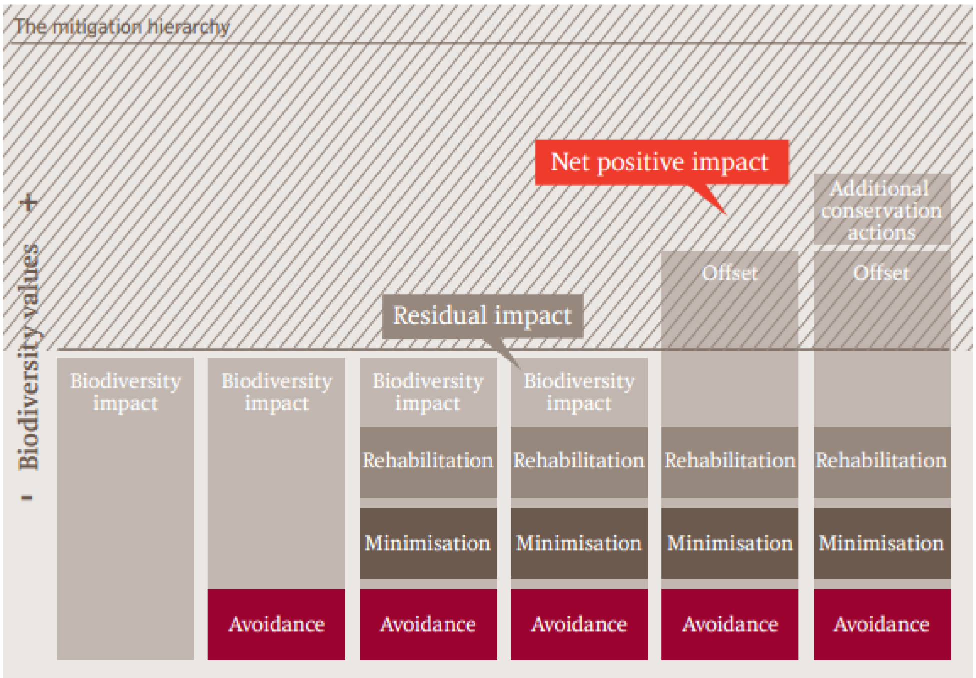

Figure 3: The mitigation hierarchy; Source: Rio Tinto

Table 2: Principles of the mitigation hierarchy; Source: (BBOP, 2012)

Principles	
Avoidance	Measures taken to avoid creating impacts from the outset, such as careful spatial or temporal placement of elements of infrastructure, in order to completely avoid impacts on certain components of biodiversity
Minimisation	Measures taken to reduce the duration, intensity and/or extent of impacts (including direct, indirect and cumulative impacts, as appropriate) that cannot be completely avoided, as far as it is practically feasible
Rehabilitation/restoration	Measures taken to rehabilitate degraded ecosystems or restore cleared ecosystems following exposure to impacts that cannot be completely avoided and/or minimised
Offset	Measures taken to compensate for any residual significant, adverse impacts that cannot be avoided, minimised and / or rehabilitated or restored, in order to achieve no net loss or a net gain of biodiversity. Offsets can take the form of positive management interventions such as restoration of degraded habitat, arrested degradation or averted risk, protecting areas where there is imminent or projected loss of biodiversity.

There can be various types of offset mechanisms, differing very slightly in their approach. An example of three offset mechanisms and their relative strengths and weaknesses can be seen in Figure 4.

Features of Compensatory Mitigation Programs Worldwide			
	Compensation Funds	One-Off Offsets	Mitigation Banking
Driver	Compliance	Compliance or Voluntary	Compliance
Policy Examples	China's Forest Revegetation Fee; Brazil's Industrial impact compensation ('developer's offsets')	Offsets under various Environmental Impact Assessment laws	US Compensatory Mitigation (aka wetland mitigation); BioBanking in New South Wales, Australia
Implementation Complexity	Low	Medium	High
Required Market Infrastructure	Low	Low to medium	High
Broad-Scale or Strategic Conservation	Dependent on program design	Less likely	More likely
Ecological Effectiveness	Dependent on design and enforcement	Dependent on design and enforcement	Dependent on design and enforcement
Who supplies the compensation?	Government	The developer	Third-party, government, or the developer
Transparency	Moderately likely	Less likely	More likely

Figure 4: Three examples of offsetting mechanisms; Source: (Madsen, et al., 2010)

The commissioning of a number of reports have peaked interest in biodiversity offsetting as a protection measure, especially in the EU; firstly and most importantly TEEB (TEEB, 2010), eftec and the IEEP's technical report for the EC (eftec, IEEP, et. al, 2010), and finally Treweek Consultants' scoping report for Defra (Treweek, et al., 2009). Defra even state on their website:

"We think that biodiversity offsetting has the potential to deliver planning policy requirements for compensation for biodiversity loss in a more effective way" (Defra, 2012)

This is slightly confusing as it is not stated what biodiversity offsetting is "more effective than", but it can be assumed to be the status quo.

Furthermore, BBOP released their biodiversity offsetting guidelines and standards in January of this year (BBOP, 2012), with the intention of refining the measures alongside a public consultation in 2013 for re-release in 2014.

1.6 Research objective

Economists and financial analysts alike are starting to take a real interest in the developments of biodiversity offsetting. A report by economists at the RSPB states:

"A strong biodiversity offset market has the potential to reduce environmental damage from development, simplify the planning system and increase funding for conservation. Likely funding raised is £53 million a year" (Comerford, et al., 2010, p. v)

Peter Carter, head of the sustainable development unit at the EIB, has been quoted as saying the biodiversity market could be "as big as the carbon market" (Levitt, 2010). This is exactly what this thesis is concerned with. The main objective of this research is to establish whether the success of the carbon market foreshadows a globally successful biodiversity market. A secondary objective of this research is to assess the potential and demand for a biodiversity disclosure project, based on the template of the highly successful CDP.

1.7 Rationale for the study

The rationale for this study is to assess whether the heavy expectation laid on biodiversity offsetting and mitigation banking is justified. The implementation of a regulatory mitigation banking scheme is a large undertaking, and policy makers need to have a comprehensive understanding of the potential of such a scheme in order to make a decision. This study is intended to lie alongside the current glut of scoping and review reports written for governments on the potential of a biodiversity offsetting scheme (Bishop, et al., 2008; Treweek, et al., 2009; BirdLife International, 2010; eftec, IEEP, et. al, 2010; Comerford, et al., 2010; Madsen, et al., 2010) as a specific comparison to the carbon market.

Biodiversity has traditionally been given none or very little value and it follows logically that in the current biodiversity crisis we find ourselves in, *any* attempt to monetise biodiversity is a step in the right direction; as the Master of Ceremonies sings in *Cabaret*: "money makes the world go round".

1.8 Outline of study

This research is divided into six chapters:

1. Introduction
2. Methodology
3. The rise (and fall?) of carbon
4. Biodiversity markets worldwide: successes and failures
5. The future of biodiversity offsetting
6. Conclusions

CHAPTER TWO

2 METHODOLOGY

This literature review can be broadly categorised into four sections (summarised in Figure 5), and was compiled solely using secondary data sources:

- Introduce the concept of biodiversity and provide a **background** to the need for a market-based conservation tool in the form of biodiversity offsetting
- **Review** the current state of both the carbon and biodiversity markets by selecting specific case studies, assess their relative success, and identify factors which contributed to this success for carbon
- Apply these success factors to the biodiversity markets, and **analyse** whether they can contribute in the same way to the development of a global biodiversity market, especially with regards to the potential for a biodiversity disclosure project
- Finally, draw appropriate **conclusions** as to the future of biodiversity offsetting in contributing to a global biodiversity market of comparable size to the modern carbon market

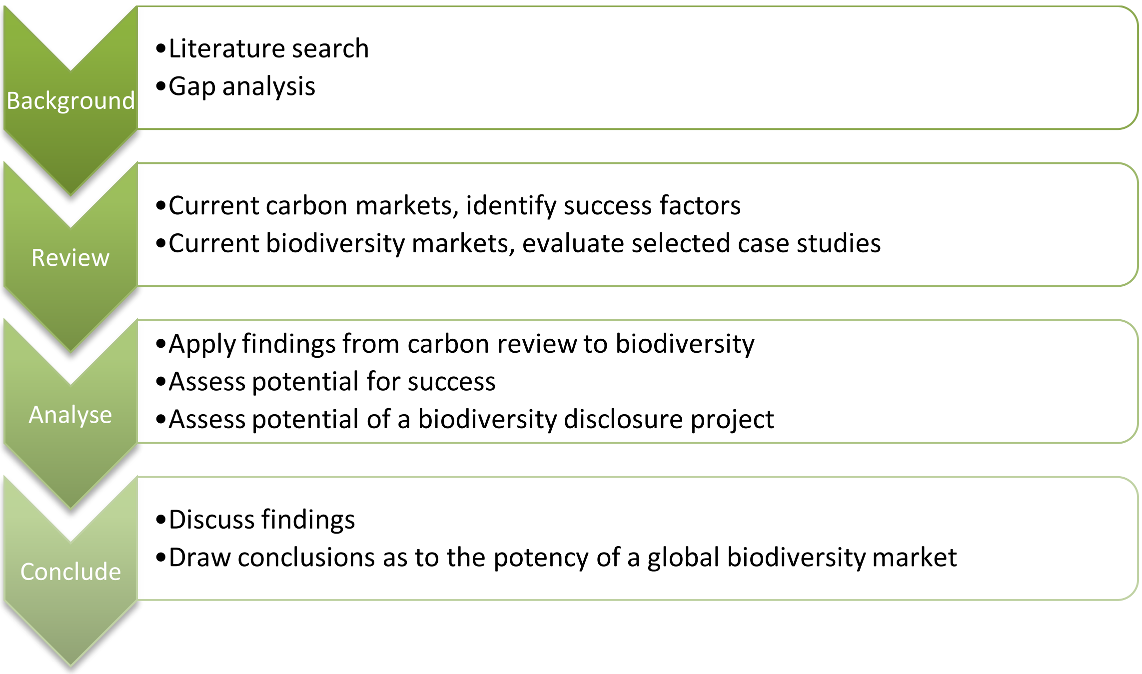

Figure 5: The four stages of this research

CHAPTER THREE

3 THE RISE (AND FALL?) OF CARBON

3.1 A brief history of carbon

CO_2 is part of the natural carbon cycle in operation since life began over 3.5 billion years ago. G. S. Callendar, a steam engineer, laid the foundation for subsequent research into carbon emissions. He was the first person to identify a potential problem with the amount of carbon being emitted post-industrial revolution in a tentatively offered research paper linking high CO_2 levels with increased temperatures (Callendar, 1938). The paper wasn't very scientifically robust, and the topic was largely ignored until a 1979 report by the National Academy of Sciences was released, urging immediate action on global warming caused by CO_2e (National Academy of Sciences, 1979).

The movement against carbon emissions slowly snowballed until the Kyoto Protocol in 1997 attempted to follow the great success of the Montreal Protocol of 1989 in fighting ozone depletion. The protocol introduced 'flexible' market mechanisms to curb emissions: emissions trading, joint implementation (investing in emissions reduction projects in other Annex I countries), and the clean development mechanism (investing in emissions reduction projects in non-Annex I countries)[1]. The EU ETS started in 2005, when Kyoto came into force.

By 2007, it became widely accepted that "global atmospheric concentrations of CO_2, CH_4 and N_2O have increased markedly as a result of human activities since 1750"; the evidence for climate change as a result of GHGs was "unequivocal" (IPCC, 2007, p. 37) (Figure 6).

[1] For a list of Annex I and non-Annex I countries, see (UNFCCC, 2012)

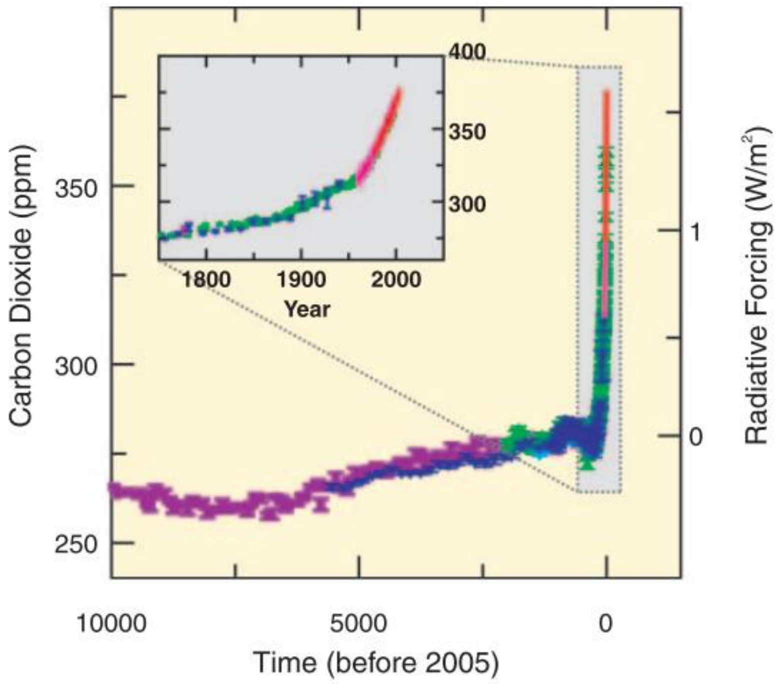

Figure 6: Changes in CO_2 from ice core and modern data; adapted from (IPCC, 2007)

The Stern Review, drafted for the UK government in 2007, made clear the financial risks involved with ignoring the threat of climate change induced by anthropogenic GHG emissions, advocating early action as the most economic response. The review estimates that if carbon emissions are not reduced by at least 25% below current levels, then the GDP will have decreased by 5-10% of what it would have been (Stern, 2007). Most importantly, the report conveyed climate change as a tangible economic risk to both businesses and consumers.

COP17 in Durban in 2012, whilst not meeting the expectations of many environmental NGOs, did ensure parties agreed to a second commitment period of the Kyoto Protocol (excluding Canada, who renounced it in December 2011), as well as the introduction of the Green Climate Fund to aid climate change financing.

The latest development in the UK carbon industry has been the introduction of mandatory carbon emissions reporting for companies listed on the London Stock Exchange, which comes into force April 2013 (Defra, 2012). This has been followed by the announcement that the EU and Australia emissions markets may be linked by 2015 – the EU carbon price will provide a price floor for the Australian market, which previously stood at an unrealistically high, and highly criticised, AUS$15/tonne (Rourke, 2012).

3.2 Carbon offsetting

The World Resources Institute (WRI) defines carbon offsetting as follows:

"A greenhouse gas (GHG) or "carbon" offset is a unit of carbon dioxide-equivalent (CO_2e) that is reduced, avoided or sequestered to compensate for emissions elsewhere" (Goodward, 2010)

There are two broad categories of carbon offsets: mandatory, or compliance, and voluntary.

3.2.1 Compliance

Carbon offsetting can be performed in compliance with 'cap and trade' regulatory mechanisms set by institutional authorities. The best known example of this is the EU ETS, which started in 2005 and is currently coming to the end of Phase II, ready for Phase III in 2013. An institutional authority (e.g. the EC) sets the 'cap' for emissions, i.e. the maximum amount of emissions allowed. A limited number of credits (permissions to emit a certain amount of carbon) are then made available that add to this 'cap'. Institutions in the scheme can then 'trade' credits according to need; heavy emitters will look to buy extra credits, whilst those with lower emissions can sell their extra credits.

3.2.2 Voluntary

Carbon can also be offset either completely voluntarily or pre-compliance through the private sector. Companies providing voluntary carbon offsetting either by buying regulatory credits to 'retire' them (i.e. reduce the number of credits available on the compliance market), or offer their own certified carbon credits. According to Ecosystem Marketplace & Bloomberg New Energy Finance, the top two ranked reasons for voluntarily offsetting were unsurprisingly CSR (32% market share) and PR/branding (22%) (Peters-Stanley & Hamilton, 2012).

3.3 The carbon market today

	2010		2011	
	Volume ($MtCO_2e$)	Value (US$ million)	Volume ($MtCO_2e$)	Value (US$ million)
Allowances market				
EUA	6,789	133,598	7,853	147,848
AAU	62	626	47	318
RMU	-	-	4	12
NZU	7	101	27	351
RGGI	210	458	120	249
CCA	-	-	4	63
Others	94	151	26	40
Subtotal	**7,162**	**134,935**	**8,081**	**148,881**
Spot & Secondary offset market				
sCER	1,260	20,453	1,734	22,333
sERU	6	94	76	780
Others	10	90	12	137
Subtotal	**1,275**	**20,637**	**1,822**	**23,250**
Forward (primary) project-based transactions				
pCER pre-2013	124	1,458	91	990
pCER post-2012	100	1,217	173	1,990
pERU	41	530	28	339
Voluntary market	69	414	87	569
Subtotal	**334**	**3,620**	**378**	**3,889**
TOTAL	**8,772**	**159,191**	**10,281**	**176,020**

Sources: World Bank, Forest Trends-Ecosystem Marketplace for data on the voluntary market and Thomson Reuters Point Carbon for data on the California offsets
Subtotals and totals may not add up due to rounding

Figure 7: Carbon market at a glance, volumes and values, calendar 2010-2011; Source: (Kossoy & Guigon, 2012)

From the small beginnings of voluntary offsets in the late 80s, the carbon market has matured into a sizeable worldwide market, even resisting the negative influence of the current global financial crisis. Recently, due to a ‘perfect storm’ combination of events (the aftermath of Fukushima on the nuclear industry, the continued debt crisis, and the downgrading of the USA’s credit rating from AAA), the price of carbon has plummeted. Despite this, the overall market size has continued to increase, growing by 11% year on year in 2010-2011 to US$176 billion, with transaction volumes reaching a new high of 10,281 $MtCO_2e$ (Figure 7) (Kossoy & Guigon, 2012). However, to place this figure in perspective: Exxon Mobil, just *one* player in the global oil industry, has a current market value of US$408.19 billion (Yahoo Finance, 2012).

There are now a variety of examples of emissions trading schemes (Table 3), whilst in the voluntary offsets market, there are over 150 retailers worldwide (Lovell, et al., 2009).

Table 3: International examples of emissions trading; adapted from (New Zealand Government, 2012)

Country	Scheme
Australia	Carbon Pricing Mechanism, nationwide cap and trade scheme by 2015
European Union	European Union Emissions Trading Scheme (EU ETS)
New Zealand	New Zealand Emissions Trading Scheme (NZ ETS)
China	Scheduled to launch pilot schemes in six provinces and cities in 2013 with a view to develop a nationwide scheme by 2015
Republic of Korea	Scheduled to begin 2015
United Kingdom	CRC Energy Efficiency Scheme
United States	California – to introduce cap and trade scheme this year
	Regional Greenhouse Gas Initiative (RGGI)
	Western Climate Initiative (WCI)

3.4 Has carbon offsetting worked?

Carbon offsetting, like any major policy, has attracted both generous praise and fierce criticism. In a general context, the words of Kossoy & Guigon accurately sum up the larger impact of carbon offsetting:

"More broadly, low-carbon initiatives, including market mechanisms, have broken the inertia and significantly raised awareness of the climate challenge" (Kossoy & Guigon, 2012, p. 11)

This viewpoint is supported by Ellerman and colleagues in their book 'Pricing Carbon'; they assert that the EU ETS has instilled a profound change in attitude and practices amongst participating firms just by being attempted (Ellerman, et al., 2010).

3.4.1 Successes

Despite the criticisms and problems, there is evidence to suggest that the EU ETS has proved that a large cap and trade emissions scheme can work. A study found that even in the extremely problematic Phase I of the EU ETS, and despite the carbon price crash c.2007 (Figure 8), emissions were reduced by 2-5% against business as usual scenarios (amounting to 120-300 MtCO_2e) (Ellerman, et al., 2010). Furthermore, it has been estimated that the EU has achieved its environmental goals through the ETS using just

1% of its GDP, and if credits were auctioned it might even be able to have a *positive* economic impact (Grubb, et al., 2009).

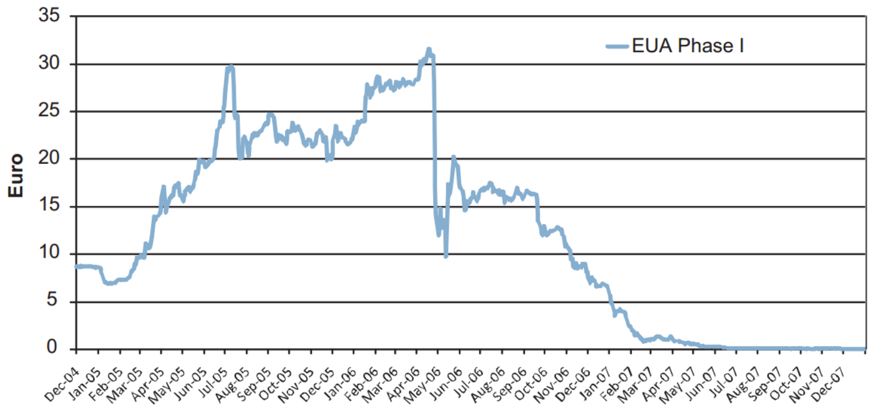

Figure 8: The carbon price crash of 2007 – price per tonne of carbon; Source: (Hintermann, 2010)

Kossoy & Guigon argue that amongst the critics and the naysayers, it is important to reflect on the cumulative positive impact of carbon emissions trading to date: pre-2013 CERs contracted forward have supported additional investments of US$130 billion in developing countries (Kossoy & Guigon, 2012).

3.4.2 Criticisms and challenges

3.4.2.1 Historic emissions

One of the first criticisms of the Kyoto Protocol was the establishment of 1990 as the base year for emissions reductions. This was seen as a slap in the face for developing countries, especially as rising CO_2 emissions had begun during the industrial revolution of 1750-1850. This essentially placed every country on a level playing field, with no regard for the cumulative effect of developed countries' substantial emissions pre-1990. This took a further hit in December 2011 when Canada renounced the Kyoto Protocol.

3.4.2.2 Environmental colonialism

The Centre for Science and Environment (CSE) in India uses this as the main base for their scathing criticisms of the carbon trading system. The centre has argued that there should be a moral difference between 'survival' emissions, e.g. cook stove emissions, and 'luxury' emissions, e.g. high consumption cars. Without this crucial distinction, the developed north is able to maintain high levels of consumption by paying next to nothing to grow trees through ill-thought out projects in developing countries, or for being awarded credits for low-hanging fruit (Lomborg, 2001) industrial solutions domestically (Liverman, 2009).

3.4.2.3 Extant emissions

UBS released a report to investors in 2011 claiming that the EU ETS had wasted US$277 billion on almost zero impact reducing emissions. The report also claimed that were these finances directed at the EU's most polluting factories, emissions could have been reduced by 40%. The report was never made public, and goes directly against other studies claiming the opposite (mainly (Ellerman, et al., 2010)). For a concise summary of the other main arguments against carbon trading, see (Kill, et al., 2010).

3.4.2.4 Voluntary offsets

The voluntary offsets sector has attracted the lion's share of criticisms, mainly revolving around the quality and efficacy of the offsets provided by the numerous offset retailers. Davies summed up public opinion well when he wrote:

"It was adopted by the corporate lobby at the Kyoto summit in 1997 and has grown into a large but deeply troubled adolescent – confused, unpredictable, and difficult to trust." (Davies, 2007)

For a good review of the main criticisms of voluntary carbon offsetting, see (Kollmuss, et al., 2008).

3.5 Why has the carbon market succeeded?

3.5.1 Homogeneity of emissions

One of the main reasons for the success of the carbon market is the homogeneity of emissions. CO_2 is a 'well-mixed gas' as a result of the turbulence of the atmosphere, and is found at all locations and altitudes. Furthermore, the potential impacts of carbon emissions are very much a global problem – the retreat of the Arctic land-based ice and the warming of the climate will have disastrous consequences all over the globe. In this way, carbon emissions foster an attitude that 'we're all in this together' and as such encourages collective action. This leads to one of the main assertions about carbon emissions, and has been used as support for the selection of 1990 as the base year: it doesn't matter where emissions are reduced, so long as they are reduced.

Arguments against this assertion focus on CO_2 as a component of air pollution in general (Jacobson, 2010). CO_2 is undoubtedly still a local as well as global pollutant, and there have been tragic cases of CO_2 poisoning (e.g. Lake Nyos in 1986). However, the consequences of global emissions far outweigh problems associated with local pollution, which can be seen as one possible reason for the growth of today's carbon market.

3.5.2 High volume of carbon research

There would not be life on Earth were it not for the organic properties of carbon. Due to its central importance, mankind have a vested interest in its research. As a result, there have been vast volumes of research dedicated to carbon – whether it be in the field of chemistry, geology, or biology. The importance of carbon and its many uses have led inevitably to a comprehensive understanding of its functioning and properties. This means that whilst carbon emissions markets may be hard to measure, they are based on the solid foundation of organic chemistry and intensive research.

3.5.3 Easily quantifiable

As carbon emissions are part of the quantitative world of organic chemistry, at the most basic level it is easy to calculate the amount of carbon released when a fossil fuel is burnt using molar concentrations. The burning of a hydrocarbon (fossil fuels) at its simplest can be summarised in the equation below:

$$C_xH_y + O_2 \rightarrow CO_2 + H_2O$$

Carbon emissions calculations from industrial installations are just more complex extensions of this basic formula, and the calculation of supply chain emissions a yet further complex version of this basic foundation.

This works for emissions reductions as well, just in reverse. A new technology can reduce the carbon emitted by a certain amount, so the overall emissions are reduced accordingly.

When it was discovered that CO_2 was not the only dangerous GHG, the quantitative nature of chemistry even allowed the formation of a standardised unit of GHG emissions, based on the functional equivalent concentration of CO_2: carbon dioxide equivalent, or CO_2e.

It is this quantitative nature that has gelled so well with financial markets and allowed the development of carbon to its current market value; carbon is truly a fungible commodity and as such functions well as an environmental currency, able to be speculated and accumulated (Salzman & Ruhl, 2000).

3.5.4 Timing

Part of the success of the carbon market can be attributed simply to fortuitous timing; the concept of carbon offsetting and concern for emissions came during a period of continued economic growth (Figure 9 – with the exception of 1993 for Europe).

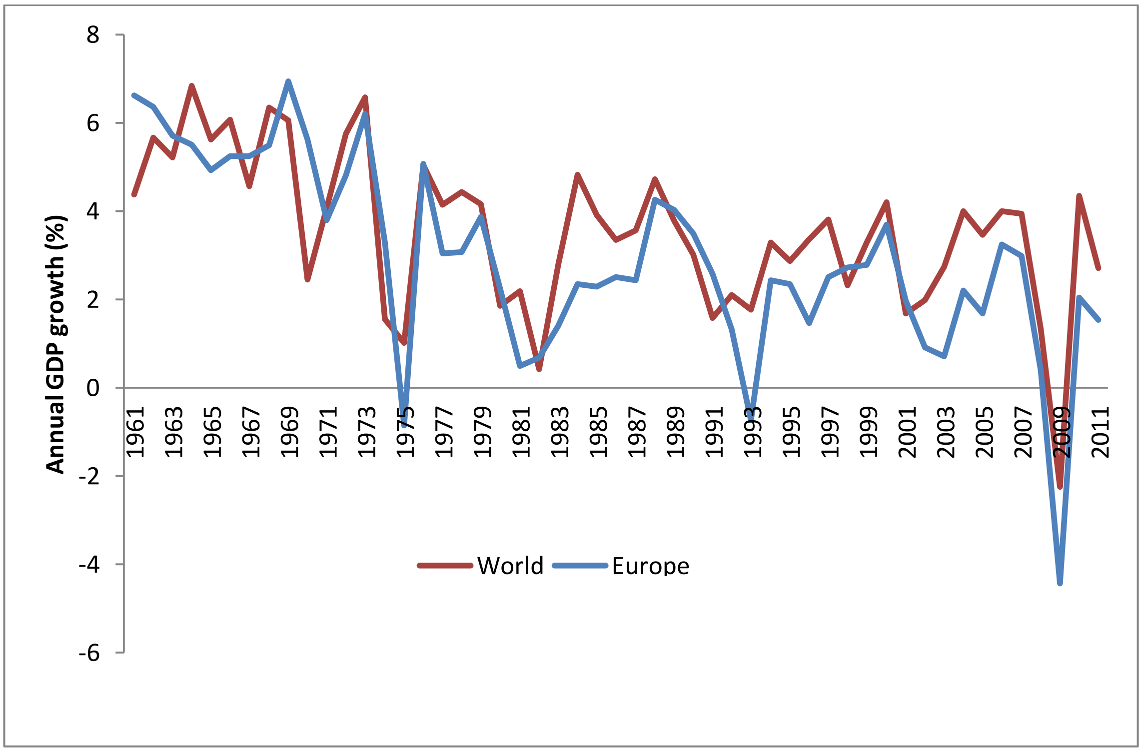

Figure 9: Graph showing world and European annual GDP growth for the period 1961-2011; Data: World Bank

This is especially true of the formation of the EU ETS. The crucial implementation of the first phase (2005-2007) spanned a period of c.1-3% annual GDP growth. This continued growth allowed for the largest component of the worldwide carbon market value to expand, as industries needed to maintain their levels of production, and carbon became established as a tradable commodity. Fortunately, this enabled the EU ETS to become properly established before the global economic downturn post-2008.

3.5.5 Offset location

The options available for carbon offsetting locations may be another reason for the growth of the carbon market. As aforementioned in 3.1, the Kyoto Protocol set out three main options for emissions reductions:

i. Emissions trading
ii. Joint Implementation (JI), where Annex I countries can coordinate with other Annex I countries to implement emissions reduction projects. These projects culminate in the production of Emission Reduction Units (ERUs)
iii. Clean Development Mechanism (CDM), where an Annex I country can comply with emissions targets by sourcing Certified Emissions Reductions (CERs) from non-Annex I countries

This flexibility of location is intended to help "disperse the effect of emission constraints globally, allowing industrialised countries (and their companies) to invest in emission reductions wherever it is cheapest globally" (Grubb, 2003, p. 159). In this way, Member States have a wealth of options available to them for assuring compliance.

3.6 The Carbon Disclosure Project (CDP)

The Carbon Disclosure Project (CDP) is a non-profit entity that launched in 2000 with the aim of side-stepping national interests to drive down GHG emissions by directly targeting large corporations. The CDP annually sends large corporations a questionnaire requesting information in four main sections:

1. Asking firms to identify the risks and/or opportunities climate change may bring
2. The current and anticipated responses of the firm to these risks and opportunities
3. Detailed emissions accounts
4. Who in the firm is responsible for climate change

The first letter requesting information was backed by 35 institutional investors representing US$4.1 trillion in assets (Stanny & Ely, 2008); the latest by more than 655 institutional investors representing US$78 trillion in assets (CDP, 2012). The response rate from the corporations is now up to 81% (CDP, 2011), showing the informational power of the CDP.

Numerous studies have shown that the more publically scrutinised a company is, the higher the costs of not disclosing information and advocating transparency (for a review see (Verrecchia, 2001)). Stanny & Ely investigated this further, concluding that the larger the firm, or the more global it is (based on volume of foreign sales), or whether the firm has previously responded to a disclosure request, the more likely the firm is to disclose (Stanny & Ely, 2008).

CHAPTER FOUR

4 BIODIVERSITY MARKETS WORLDWIDE: SUCCESSES AND FAILURES

Ecosystem Marketplace's 2010 report on The State of Biodiversity Markets estimated the global market size at US$1.8-$2.9 billion as a lower limit, as 80% of markets are not transparent enough to accurately estimate their size (Madsen, et al., 2010). The report found 39 active programmes worldwide, with 25 in development, and over 86,000 hectares either protected or restored. Whilst this is still not close to the current carbon market size of US$176 billion (Kossoy & Guigon, 2012), the 2011 Update to the State of the Biodiversity Markets report estimated the market size at US$2.4-$4.0 billion, with 45 active programmes, 27 in development, and over 187,000 hectares protected or restored (Madsen, et al., 2011). This demonstrates an impressive market growth of 33-38% in just a year, with a 117% increase in land either protected or restored.

As Ecosystem Marketplace themselves admit, estimating the state of the global biodiversity market is challenging because biodiversity markets are "hard to define, fragmented, swiftly changing, and opaque" (Madsen, et al., 2010, p. vii). Programmes vary from fully developed schemes with active mitigation banking to programmes channelling development impact fees towards one-off offsets.

When assessing the success of any one biodiversity programme, Kentula warns against the confusion of different types of success. Lewis originally defined success with regards to wetlands mitigation as the "achievement of established goals" (Lewis, 1990), but Kentula breaks this down into component parts (Table 4).

Table 4: Types of success involved in wetland mitigation; Source: (Kentula, 2000)

Type of success	Definition
Compliance success	Determined by evaluating compliance with the terms of an agreement
Functional success	Determined by evaluating whether the ecological systems of the system have been restored
Landscape success	A measure of how restoration (or management, in general) has contributed to the ecological integrity of the region or landscape and to achievement of goals such as the maintenance of biodiversity

Whilst Kentula offered these definitions for wetland mitigation, they can be broadly applied to regulatory compensatory mitigation in general, and the concept of breaking down 'success' into component parts is an important one. Furthermore, the idea that compensatory mitigation may transcend simple yes/no successes is also important; success in these projects needs to be a continuing measurement towards (sometimes shifting) goals and objectives.

4.1 Mitigation and conservation banking in the US

In much a similar way as the EU ETS dominates the global carbon market, the global biodiversity market is dominated by the U.S. compensatory mitigation and conservation banking schemes, which generate US$2.0-$3.4 billion per annum.

4.1.1 Wetland mitigation banking

The U.S. has a long history of compensatory offsets, with wetland mitigation starting in the 1970s. The scheme has matured into an impressive example of what a compliance biodiversity offsetting system can become with the right backing, as the market is driven by compliance to the Clean Water Act (§404) of 1972 and the principle of 'no net loss'. The system also has considerable institutional infrastructure. Ecosystem Marketplace describes the scheme as "the most market-like offset program in the world" (Madsen, et al., 2010, p. 8), with its own price signals, third party involvement and credit units standardised enough to *trade* (although this will be discussed further below). Private actor participation allows local environmental entrepreneurs to create and sell environmental permits for profit.

In the programme, a permittee (usually commercial or residential developers, who make up one third of the demand for credits (Madsen, et al., 2010), or a government infrastructure developer) has three options after following the mitigation hierarchy (Figure 3):

- Create their own offset (called permittee-responsible mitigation)
- Pay towards in-lieu fee (ILF) programmes
- Pay via third party mitigation banks

Indirect offsets (e.g. payments to fund research into wetlands) are not allowed. Furthermore, offsets must be located in the same watershed as the wetland damage.

The wetland mitigation programme is undoubtedly a large, complex system and as such, much like the EU ETS, relies on an iterative learning process. New regulations were introduced in 2008 in an attempt to combat some of the more pressing criticisms: payment via a third party mitigation bank is now first preference, followed by ILF and permittee-responsible mitigation (to combat 60% of mitigation still coming from permittees as a result of more stringent regulations for ILFs and mitigation banks); this is supported by an attempt to introduce equivalent standards for all three. Anyone selling credits will have long-term funding requirements.

4.1.1.1 Transparency

The main problem with the wetlands mitigation banking programme is transparency, and is a result of the scale of the operation. Transparency is one of the pillars of modern economics, and is a requirement if a market is to approach an efficient state, as any market mechanism is designed to do. The new 2008 regulations have attempted to address this issue, but problems remain.

- **Credits**: The national range of wetland credits is US$3,000-$653,400; this is encouraging and indicates that, generally, wetland permits in Virginia (US$653,400) are rarer than wetland permits in Arkansas (US$3000) (Madsen, et al., 2010). However, credits are heavily regionally dependent. Each region decides on the wetland classification system it uses; this is most commonly based on Cowardin and colleagues' system (Cowardin, et al., 1979), where wetland types are further sub-classified according to associated flora, but not always. Other credits are issued on the type of method used to create them, whether this be restoration, enhancement, creation, etc. Furthermore, credit calculation methods are also designed on a regional level, and are a specific combination of acreage, functional assessment, and best professional judgement, dependent on the region. As such, credits cannot be compared nationally like-for-like, making it hard to analyse the national market and draw conclusions. However, this does help to reinforce the idea that biodiversity offsets, unlike carbon, should be made locally.

- **Mitigation banks**: As a result of transparency criticisms, the U.S. Army Corps of Engineers (USACE) have improved their Regulatory and In-lieu fee and Bank Information Tracking System (RIBITS) dramatically. Ironically, this has only served to highlight the ambiguity in the status of mitigation banks. This can be seen in the jump in the number of active mitigation banks from 431 in 2009 to 789 in 2010 (an 85% increase) and reduction of 'unknown' mitigation banks from 60 in 2009 to 32 in 2010 (down 47%) (Madsen, et al., 2011). Consequently, this might hide any genuine increase in the number of mitigation banks due to the 2008 regulations, which provide regulatory certainty for bank approval. Increased transparency would aid research into the scheme as a whole, which may in turn provide positive iterations to the policy.

4.1.1.2 Compliance success

Despite being set up as a regulatory instrument driven by compliance to the Clean Water Act, wetlands mitigation in the U.S. cannot be seen as a compliance success. It has been suggested that non-compliance has led to the failure to meet the no net loss policy; in eight studies, non-compliance led to an average of just 0.69 hectares of wetlands implemented for every hectare lost (Turner, et al., 2001). This is echoed by Ambrose, who concedes that "Few mitigation projects are in compliance with all of their permit conditions" (Ambrose, 2000, p. 2).

The two principal requirements for a permit are 75% hydrophytic plant cover by two growing seasons and the establishment of an area the same size as the wetland being impacted. A study based in Massachusetts found that 54.4% of projects were non-compliant with regulations for a variety of reasons. However, the study found that if projects were simply built whenever they are permitted and to the same size as the impacted wetlands (i.e. removing these two causes of failure), compliance frequencies would increase from 41.2% to 83.3% (Brown & Veneman, 2001) .

A large factor causing non-compliance are the goals and objectives set by the USACE. Compliance rates have been found to double when a permit contained more specific conditions to be met; furthermore, compliance rose to 100% from 50% when a permit contained a deadline to meet (Turner, et al., 2001). Worryingly, the same study asserted that “Compliance monitoring is known to be nonexistent after five years” (Turner, et al., 2001, p. 15).

Matthew & Endress confirmed this compliance dependency on the goals and objectives set; they even argue that some of the standard goals set are too lenient to be of any use. They call for more appropriate goals to be set on a case by case basis, based on reference to either previous similar restoration projects or natural wetlands (Matthews & Endress, 2008).

Significant variations in compliance frequencies across states and projects make it hard to establish the overall compliance success of the wetlands compensatory mitigation programme, but scientists seem to agree on worrying levels of non-compliance.

4.1.1.3 Functional success

The no net loss policy of the programme extends to no net loss of wetland *function*. Brown and Lant concluded that:

“the concept of wetland mitigation banking is a sound one, so long as it is recognized that a spatial redistribution of wetlands, and therefore wetland functions and ecosystem service values, is inevitable through the operation of banks” (Brown & Lant, 1999, p. 344)

However, this makes the assumption that wetland function can be artificially created, and this goes against a strong tide of scientific opinion.

Bendor found that delays in initiating and completing wetland mitigation projects can contribute to a consistent and considerable net functional loss (Bendor, 2009). Not only are delays in implementing wetland projects fuelling non-compliance, but they are also reducing the likelihood of achieving no net loss of function.

The main problem with restoring functionality in U.S. wetlands is time; as Mitsch and colleagues conclude: a fully functioning wetland will come about "in its own time, not in ours" (Mitsch, et al., 2012, p. 250). A study in Pennsylvania found that created and natural wetlands had significant differences in soil and vegetation structure. The authors concluded that a stronger effort is needed to recreate native wetland species, but more importantly they called for a more careful consideration of monitoring period lengths (Campbell, et al., 2002). A follow up study in Pennsylvania again found significant differences in soil and vegetation structure, even for created wetlands more than 12 years old. The study concluded that the created wetlands studied have still not met the goal of no net loss of function, and do not even appear to be on the right successional trajectory to do so (Hoeltje & Cole, 2009).

However, functionality studies are very much constrained by current methodologies. Functionality studies most commonly use hydrogeomorphic (HGM) models. These are *very* limited; HGM models do not even directly measure *function*, but *functional capacity*. The models use physical traits of the wetlands to predict the ecosystem's functional capacity, using the assumption that natural reference wetlands will be operating at full functional capacity (Hoeltje & Cole, 2009).

Even if wetland scientists had a perfect methodology for assessing functionality, Zedler outlines the difficulties restorationists would still face (Zedler, 2000). Ecological succession is an extremely complex and unpredictable process; every management decision made can shift an entire ecosystem onto a completely different successional trajectory. Zedler is not pessimistic though, and simply advocates more experimental restoration projects. Like the EU ETS, the U.S. wetland mitigation programme is an *iterative* process.

This year, the results from a 15 year wetland study were released by Mitch and colleagues, asking if even 15 years is enough to measure functionality against natural reference wetlands (Mitsch, et al., 2012). The study compared two identical sites with identical hydrologic inflows. One was planted and one was not. After 15 years, the planted site had a higher community diversity, but the unplanted site was functionally more productive. The study highlights the dilemma that wetland managers and policy makers face: it may be impossible to artificially imitate natural wetlands, and that policy goals should reflect that.

Regardless of functionality methodology and timeframes, the National Research Council (NRC) released a report in 2001 concluding that the U.S. had not yet achieved no net loss of function (National Research Council, 2001).

4.1.1.4 Landscape success

The landscape success of any biodiversity offsetting system must be measured in terms of the maintenance of regional and, by extension, national biodiversity. There has been significant research interest in the minimal wetland density and proximity required to maintain wetland species populations (Semlitsch & Bodie, 1998; Gibbs, 2000; Keddy, 2010). Wetland species exist in metapopulations, maintaining local populations by occasional migration. Gibbs found that the protection of all wetlands over 1 acre is required to maintain the minimum wetland density required to conserve wetland species. This seems unlikely under the wetlands mitigation programme as the policy leans towards restoring and enhancing larger wetland sites at the expense of smaller ones.

Burgin echoes the sentiments of the NRC 2001 report; whilst the wetlands compensatory mitigation banking system has not been an “unmitigated disaster”, it is at best “moderately successful”, with “the best current outcomes appear[ing] to be a slowing of the rate of biodiversity decline” (Burgin, 2010).

4.1.2 Conservation banking under the Endangered Species Act

Conservation banking in the U.S. is facilitated by the legal requirements of the Endangered Species Act (ESA) of 1973 (§7 & §10), and applies to developments affecting endangered and threatened species. The scheme is heavily based on its wetlands mitigation banking system predecessor. Developers must follow the mitigation hierarchy (Figure 3), before offsetting any residual impacts either through

- Permittee-responsible mitigation
- ILFs
- Paying a conservation bank

In contrast to the wetlands system, conservation banking has no stated policy of no net loss; this is replaced by a 'species recovery goal'. Regulation is carried out by the U.S. Fish and Wildlife Service (USFWS) or the National Marine Fisheries Service (NMFS). The system is primarily prominent in California, where regulatory duties are carried out by the California Department of Fish and Game (CA DFG) for all Californian endangered and threatened species.

Whilst there are no official regulations yet, federal agency guidance was introduced in 2003 (over a decade after conservation banking had started) to allow public and private conservation banks and ILF programmes.

Developers impacting endangered or threatened species require an authorisation (called 'incidental take') under section 7 or 10 of the ESA. The appropriate regulatory agency decides whether the impact is unavoidable and whether it can be offset. If so, the regulator then calculates how many credits are required to continue the development. As with wetlands mitigation, the main buyers of species credits are commercial and residential developers and government infrastructure developers, etc. Offsets are created in advance of the development project and primarily through preservation and management rather than creation or restoration. The USFWS has taken the view that improving the quality of habitats harbouring existing populations (increased connectivity, reduced edge effects, etc.) is preferential to creating new reserves. Offsets are required to be located within the 'Service Area' of the bank, as agreed in the bank agreement with the regulator. ‘Service Areas’ are usually defined by a species’ recovery unit, watershed, or other general geological feature.

The main advantage of the scheme through the eyes of private landowners is economic; it turns a legal liability (the endangered species) into a financial asset (the credit) (Fox & Nino-Murcia, 2005). Good reviews of the history and the potential of conservation banking as an environmental policy tool are provided by Bonnie and Bean & Dwyer (Bonnie, 1999; Bean & Dwyer, 2000).

There are currently 132 conservation banks in the U.S. (active, inactive, pending, sold out and unknown) (Madsen, et al., 2011); these facilitate total payments for conservation banking per annum (2009) of US$200 million (Madsen, et al., 2010).

- **Credits:**
  As with the wetland mitigation system, conservation banking is somewhat hindered by the range of credits provided and the regional differences. Credit units are most commonly assigned as a function of the acreage of habitat, but a credit unit can also refer to a breeding pair, or even a linear foot of riparian habitat. Where a species recovery plan exists (see (US FWS, 2012) for a list of species), some scientific evidence may indicate an appropriate area for credit calculation, but for new species, the first bank that attempts to gain credits generally sets the precedent for subsequent credits. Ecosystem Marketplace found 92 species credit types and 51 habitat credit types in 2009, with a price range of US$2,500-$300,000 (Madsen, et al., 2010). Species credits should be preferred over habitat credits (even though species and species/habitat combination credits are currently issued by only 9% of banks (Fox & Nino-Murcia, 2005)), as stochasticity problems affect depleted populations regardless of habitat size, thus it seems more logical to focus more intently on absolute species numbers (Lande, 1993).

4.2 The Biodiversity Offsets and Banking Scheme, New South Wales, Australia

Australia carries out some of the most advanced research into market-like mechanisms for biodiversity conservation in the world; Ecosystem Marketplace identifies it as "fertile ground" for a nationwide biodiversity offsetting scheme for three main reasons (Madsen, et al., 2010, p. 47):

- There is a general acceptance of market-like instruments for biodiversity conservation
- Australia has some highly endangered and endemic biodiversity
- There is a wealth of biological data and research capacity

However, the report also warns that nationwide implementation may also be hampered by some legal issues, namely:

- Most offsets are permanently protected, but with insufficient funds for long-term management; this poses a problem in a country with significant issues with invasive species, and management is a constant requirement
- The majority of rural Australia is 'leasehold land' where permanent protection cannot occur
- Offsets might be in direct competition with other incentive schemes, providing no environmental gain, or additionality, over what is already being carried out on the land
- The mining industry is so strong in Australia that some mining legislation takes precedent over all other legislation, almost negating the impact of offsets on the extractive industry

Regardless of these issues, the New South Wales (NSW) Biodiversity Offsets and Banking Scheme (BioBanking) was developed in 2007 off the back of several pieces of legislation: the Environmental Planning and Assessment Act of 1979 (NSW), the Threatened Species Conservation Act of 1995 (NSW), and the Threatened Species Conservation (Biodiversity Banking) Regulation of 2008 (NSW). Regulatory responsibilities are carried out by the NSW Department of Environment, Climate

Change and Water (NSW DECCW), now known as the NSW Office of Environment and Heritage (NSW OEH) as of April 2011.

The scheme existed as a pilot project until autumn 2009 when it went live. It is driven by regulatory requirements to offset impacts from urban development; important in a country where 40% of nationally listed threatened ecological communities are present in urban areas (Burgin, 2008).

Developers can voluntarily use the system to mitigate and offset their impacts, but must pass an 'improve-or-maintain' test requiring strict adherence to the mitigation hierarchy (Figure 3) before joining the scheme.

Unlike the U.S. wetlands compensatory mitigation and conservation banking schemes, NSW BioBanking has its own centralised BioBanking Assessment Methodology and associated Credit Calculator software, as well as the BioBanking Trust Fund into which all payments from developers must be paid.

As of 2011, the BioBanking scheme encompasses five BioBanks covering a total of 210.3 hectares (Madsen, et al., 2011).

4.2.1 Too early to assess success?

NSW BioBanking is very much in its infant stages and as such it is hard to determine the success to date of the scheme at the functional, compliance or landscape level. In terms of compliance, all that can be said so far is that non-compliance has been a significant hindrance to previous mitigation bank projects (Gibbons & Lindenmayer, 2007), that there are provisions in the NSW Threatened Species Conservation Amendment (Biodiversity Banking) Bill 2006 to deal with non-compliance, and that there needs to be clearer insurance against the risk of failure of BioBanks to assure potential investors (Burgin, 2008).

In terms of functional success, NSW BioBanking faces the same hurdles as any other mitigation banking programme, mainly the unpredictability of ecological succession. A recent study highlights this in an Oceanic context: artificially restored woodland in southern NSW contained fundamentally different bird assemblages than comparable natural regrowth and old growth woodland sites (Lindenmayer, et al., 2012).

4.2.2 *Economic success*

One possible way to assess the scheme's success to date is to evaluate its current economics in an attempt to predict whether the scheme will generate considerable revenue in the future.

Currently, predictions from the 2010 Ecosystem Marketplace report that high upfront costs (AUD50,000 – 60,000) may put landowners off speculating with offsets seem to be ringing true (Madsen, et al., 2010). Economic activity of the scheme has fallen below expectations; thus far 757 credits have been transferred and retired, with credit prices ranging from AUD2,563 to 8,000. The total cumulative value of credits sold is currently AUD2.8 million, with the NSW OEH's Biocertification Program projecting collection of AUD337.9 million over a 30-40 year period (Madsen, et al., 2011).

However, despite promising trading figures, demand seems to be outstripping supply; there has been a reported shortage of 22,000 ecosystem credits and 5,000 endangered species credits. Furthermore, in the Sydney basin, Cumberland plain woodland is now so highly endangered that it is in direct competition between offsetters and developers. This is coupled with increasing political pressure to look for sites outside the basin, which would compromise the conservation goals of the scheme as a whole.

Landowners are raising concerns over the high costs of the scheme's implementation: assessments using the BioBank Assessment Methodology can cost in the region of AUD25,000; although Madsen and colleagues point out that this is likely to decrease as ecologists gain experience using the methodology (Madsen, et al., 2011). Furthermore, the methodology itself has been criticised and blamed for delays in offset creation; a revised version designed to combat these criticisms is currently being formed (NSW OEH, 2011).

Despite these setbacks, Burgin concludes that:

"Unlike most legislation that aims to conserve biodiversity, there appears to be some level of acceptance by all stakeholders for BioBanking" (Burgin, 2008, p. 814)

Burgin cites the level of industry support and the foundation of previous government initiatives as the strengths of the BioBanking scheme that may translate into compliance, functional, and landscape success.

4.3 UK

4.3.1 Developments

Contemplation of the adoption of new conservation tools in the UK has been accelerated by concerns over the loss of British biodiversity (Natural England, 2008), and this includes biodiversity offsetting as a measure to balance development with conservation. When Nick Herbert first outlined a plan for mandatory biodiversity offsetting to the Labour government in 2009, the suggestion fell on deaf ears. However, the idea has gained significant traction under the current Liberal Democrat-Conservative coalition. The main reason for this is that any offsetting scheme is likely to be administered largely by local governments, which aligns with David Cameron's vision of the 'Big Society' – empowering local governments.

As of April 2012, Defra have begun piloting offset sites (

Table 5) in a programme that will run for two years to inform any decision on the development of a nationwide scheme (Defra, 2011).

Table 5: UK biodiversity offsetting pilot sites; Source: (Defra, 2012)

Location	Comments
Devon	Especially: Exeter and East Devon Growth Point, South Devon (including South Hams SAC)
Doncaster	-
Essex	Working with the Environment Bank Ltd to trial a conservation credit brokering system
Greater Norwich	-
Nottinghamshire	Likely to include the recently confirmed Humberhead Levels Nature Improvement Area
Warwickshire, Coventry and Solihull	-

4.3.2 Potential for success?

The UK, much like Australia, seems fertile ground for the introduction of a biodiversity offsets scheme. Much of this is down to extensive experience dealing with determining compensatory requirements under the EC Habitats Directive (92/43/EEC), e.g. Associated British Ports' dealings in the Humber estuary (BirdLife International, 2010). Many protected reserves in the UK are under the management of the RSPB: a large, experienced NGO who has consistently achieved net gain of biodiversity on their sites.

Furthermore, the UK conservation sector is in dire need of investment; the *Making Space for Nature* report reported a financial shortfall in UK conservation of £1 billion per annum (Lawton, et al., 2010), making the £7.5 million promised to Nature Improvement Schemes by Defra merely "start-up seed funding" (Tew, 2012). Interest in conservation banking systems has peaked further from the TEEB's findings that, were all the benefits of nature to be fully accounted for, the return on an investment in nature can be as high as 10:1 (TEEB, 2010).

The Environment Bank Ltd, a vocal proponent of habitat banking in the UK, is expectant of a successful pilot scheme in Essex for the following reasons (Tew, 2012):

- Continued levels of high economic growth with associated development programmes
- Rich natural heritage now mostly fragmented and degraded, therefore there is a high potential for restoration
- Association with the Mineral Products Association paves way for developer involvement when the scheme is scaled up

The general advantages of setting up a habitat banking scheme, as proposed by Defra, are covered by Briggs and colleagues (Briggs, et al., 2009).

Much of the success of a UK scheme depends largely on the restorability of the varied habitat types. The UK is somewhat of a special case in terms of conservation as many of its most recognisable habitats (old growth woodland, chalk grasslands, heathland, etc.) are actually only semi-natural: the result of hundreds of years of management or human disturbance. Consequently, there has been a lot of research as to what habitats can be adequately restored:

- **Saltmarshes, mudflats and freshwater reedbeds:** numerous examples of successful creation, even on relatively short timescales (Atkinson, et al., 2001; Morris, et al., 2004); RSPB has ample experience managing
- **Neutral and calcareous grassland:** grasslands that resemble ancient semi-natural grassland take a minimum of 100 years to establish (Gibson & Brown, 1991), although pseudo-functioning imitations have been made on shorter timeframes (Vecrin & Muller, 2003; Morris, et al., 2006)
- **Heathland:** extensive research into this area, with highly variable results, depending on the original state of the land
- **Woodland:** some of the 'natural' assemblages in ancient woodlands are the result of hundreds of years of management, making them essentially impossible to recreate on a policy timescale

Whilst the UK may provide a suitable atmosphere for a biodiversity offsetting scheme, the success of any scheme depends hugely on the monitoring and compliance measures applied, and Defra must be aware of this. In its current state as a completely voluntary system, UK biodiversity offsetting does not look set to become a large market.

CHAPTER FIVE

5 THE FUTURE OF BIODIVERSITY OFFSETTING

5.1 Hitting a moving target

5.1.1 Documenting biodiversity

"It is a remarkable testament to humanity's narcissism that we know the number of books in the U.S. Library of Congress on 1 February 2011 was 22,194,656, but cannot tell you—to within an order-of-magnitude—how many distinct species of plants and animals we share our world with" (May, 2011)

Conserving biodiversity is still a daunting prospect because at the most basic level, we still do not know what we have to lose. Estimates of species loss are still very much *estimates*, most likely underestimates, as they just concern the species we are aware of; some species may become extinct before their discovery.

A recent Conservation International expedition to Suriname discovered at least 46 species new to science in just three weeks, including a new type of armoured catfish and a species of frog nicknamed the 'cowboy frog' as a result of ankle spurs and white lines running down its haunches (O'Shea, et al., 2011). Worldwide, we are currently discovering species at a rate of c.15,000 per annum (May, 2011).

This is not to say that there have not been valiant attempts to estimate global species numbers: the holy grail of taxonomy. The first ballpark figure was calculated by Terry Erwin, working on specialist Coleopteran species in the tropics (Erwin, 1982); he came to the conclusion that the Earth held 30 million species of arthropods alone. This was followed by Robert May's seminal paper on the subject (May, 1988). The most recent and accurate attempt to estimate global species numbers puts our natural biodiversity capital at c.8.7 million eukaryotic species, leaving 86% of terrestrial and 91% of marine species still uncatalogued (Mora, et al., 2011). May points out that at the current rate of discovery, biological science will have completed its quest in 480 years (May, 2011).

5.1.2 The mutability of species

Biodiversity research is furthered hindered by the uncomfortable fact that species are not as immutable as we would like to think. Central to this is the concept of 'cryptic species', defined by Bickford and colleagues as:

"two or more distinct species that are erroneously classified (and hidden) under one species name" (Bickford, et al., 2007)

Their review paper emphasises the great importance of research into cryptic species complexes for conservation; a greater understanding of what habitats cryptic species are more likely to be found can help focus future research. The importance to conservation of the subject can be highlighted by four examples of species crypsis (Bickford, et al., 2007):

1. The common blue mussel, *Mytilus edulis*, is an important biological indicator species for pollution monitoring. However, the 'species' has actually been found to be a complex of three different cryptic species, all with different growth rates (Geller, 1999). This may cause irregularities in indicator data, and a consequent risk to human health.
2. Molecular work has divided two species of Southeast Asian frogs, *Odorrana livida* and *Rana chalconota*, into at least 14 different nested species (Stuart, et al., 2006). This means that two species, previously assumed to have an extensive geographic range, have been reclassified as pockets of more endemic species. This would have crucial ramifications were the geographical range to include biodiversity offset sites.
3. An integrative morphological study of rhacophorine frogs in Sri Lanka recently increased the number of species from 18 to over 100 (an increase of over 450%) (Meegaskumbura, et al., 2002)
4. Another molecular study (using mitochondrial DNA and karyotypic evidence) suggests that one of the four subspecies of the endangered northern sportive lemur, *Lepilemur septentrionalis*, is actually a distinct species (Ravaoarimanana, et al., 2004)

Conservation banking in the U.S. already has provisions to reduce the risk to landowners of previously unthreatened species being reclassified as threatened; any site must provide a comprehensive list of endangered species, as well as candidate species likely to become so in the future. However, surveys that collect this information do not have the technology or the time to account for cryptic species. If they are as widespread as Bickford and colleagues suggest, this could pose a substantial risk to landowners and the implementation of offset programmes.

5.1.3 Quantifying biodiversity

"[Biodiversity] is a fundamentally multidimensional concept: it cannot be reduced sensibly to a single number" (Purvis & Hector, 2000)

The most widely accepted and used single measure of biodiversity is species richness, but it is by no means the only measure (for a good summary, and an indication of just how complex the field is, see (Magurran, 2004)). Although multidimensional, if the components of biodiversity and the relationship between them are known, an arbirtrary number can be calculated (Purvis & Hector, 2000), but it needs to be clear that it is very much *abritrary*. This is in essence, how the credit system of biodiversity offsetting works. However, as Gotelli and Colwell point out, even measures and comparisons of species richness depend heavily on sampling effort and abundance (Gotelli & Colwell, 2001); for accuracy, data should be drawn from much more work-intensive accumulation and rarefaction curves.

Asa result of the complexity of reducing biodiversity to a single number, there have been a wealth of studies into the concept of 'barcode taxonomy' or 'DNA barcoding' (Stoeckle, 2003; Blaxter, 2004; Hebert & Gregory, 2005; Meier, et al., 2006; Hajibabaei, et al., 2007; Valentini, et al., 2009). There is even now a well established Consortium for the Barcode of Life (CBOL, www.barcodeoflife.org), which seeks to encourage and coordinate research into the subject. DNA barcoding uses a standardised DNA region (a 658 base pair region in the gene encoding the mitochondrial cytochrome *c* oxidase 1 for animals) as a tag for rapid and accurate species identification (Valentini, et al., 2009). As Valentini and colleagues point out, the concept has been around for a long time (many of the studies mentioned in 5.1.2 used the technique), but is only recently benefiting from aggregation and collaboration.

The most exciting prospect of DNA barcoding for biodiversity offsetting is its potential in rapid, inexpensive biodiversity assessment. The technique could allow for biodiversity assessment from environmental samples alone (such as soil and water). DNA barcoding technicians could perform a full assessment for as little as US$2.5-$8 per sample, depending on laboratory facilities (Valentini, et al., 2009). This kind of rapid biodiversity assessment would be incredibly useful in environments with extremely high species diversity, where it would be unreasonable to expect taxonomists to detail the biodiversity via morphology alone. Furthermore, the current set of biodiversity indices (species richness, Shannon's, Simpson's) could be supported by new indices based on 'molecular operational taxonomic units'.

Whilst it is clear that we are a long way away from stocking biodiversity assemblages like we are stacking supermarket shelves, the field of barcode taxonomy is developing rapidly and has incredibly promising prospects for biodiversity science. However, as Karl Falkenburg, EC Director-General for Environment rightly comments:

"With the carbon market we know what we are trading and how to tackle them. We can set a cap and use the price to drive them down. We have a baseline for biodiversity in Europe now, but it is not one figure - it is four pages of different elements of biodiversity" (Madsen, 2010)

Whilst carbon has been described as the "currency of the new world order" (Kelly, 2007), biodiversity fails as a currency in almost every measurable way (Salzman & Ruhl, 2000; Walker, et al., 2009).

This puts the future of biodiversity offsetting in an awkward position; unlike carbon offsetting, where the incomplete science of carbon sequestration and emissions is founded on solid atmospheric science and arguably quantifiable units (see 3.5.3), biodiversity offsetting is left as the incomplete science of complex ecosystem functioning balancing atop the incomplete science of quantifying biodiversity. Unfortunately, this makes for a very shaky scientific platform indeed.

5.2 Global crisis?

The strong case for a global biodiversity crisis is outlined in 1.4. What can be concluded is that there is undoubtedly a growing consensus forming on the current state of the world's natural capital and concern for its decline, and an increasing number of research papers are referring to the 'Holocene mass extinction' and Earth's sixth mass extinction (Barnosky, et al., 2011).

The growing concern for the decline of global biodiversity can be compared to the atmosphere regarding carbon emissions in the 1980s after the publishing of the National Academy of Sciences report (National Academy of Sciences, 1979). That atmosphere eventually led to the establishment of the IPCC in 1988, after which carbon emissions, and offsetting, began to gather momentum. An argument can be made that we are now in that position with biodiversity; the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) was established in April of this year (UNEP, 2012) with the intention of becoming "the mechanism that addresses the gaps in science policy interface on biodiversity and ecosystem services" (IPBES, 2012). From this point of view, it seems like biodiversity offsetting may be on similar tracks to carbon in the 1980s.

However, carbon managed to capitalise on the fear of a *global* crisis – the homogeneity of carbon emissions and its potential impacts (3.5.1) have fostered as much of a "we're all in this together" attitude as human nature allows. So how far can the biodiversity crisis be seen as *homogenous* in its nature and impacts?

At first, the biodiversity crisis seems very much *heterogeneous* in nature. Recent extinctions (thylacine, Steller's sea cow, dodos, passenger pigeons) cannot be said to have had anything but a local impact. However, there are examples of 'trans-boundary biodiversity', whose extinction would have a much wider impact: wildebeest and their hugely important African migration, krill as a keystone species in the rich Arctic waters, and various other migratory species. Furthermore, as pointed out by the many critics of Costanza and colleagues paper (Costanza, et al., 1997), mainly Toman (Toman, 1998), impacts on biodiversity are homogeneous because our entire built economy depends on it (Bishop, et al., 2008).

When comparing carbon and biodiversity, it is crucial to remember that the two are not by any means mutually exclusive; biodiversity is likely to follow carbon's trajectory because the two are very much intertwined. The most pressing example of the relationship is coincidentally an example of a homogeneous, global biodiversity decline: the spread of chytridiomycosis in amphibian species as a result of *Batrachochytrium dendrobatidis* (*Bd*) infection. Almost a third of amphibian species are considered threatened, with 43% experiencing some sort of population decline, and it is thought that many are down to the spread of *Bd* accelerated by anthropogenic climate change (Rohr, et al., 2011). Furthermore, to expand to biodiversity as a whole, modelling on climate predictions has concluded that anthropogenic climate change has left 15-37% of species globally 'committed to extinction' (Thomas, et al., 2004).

Although biodiversity being in bed with carbon can be seen as a factor contributing to the almost inevitable rise of a substantial biodiversity market, it can also be seen as a hindrance. The decline in global biodiversity may be seen as just one impact of climate change, and not a crisis in its own right, diverting effort away from conservation (and conservation banking) towards mitigating climate change. However, the establishment of the IPBES should help with this distinction.

5.3 Volume of biodiversity research

Research into biodiversity has been on the rise, but it is clear that there are still large gaps in our knowledge – as alluded to in 1.2.4 and 5.1.2, more research is needed into the relationship between biodiversity and ecosystem functioning, as well as cryptic species complexes and their distribution.

However, a simple search with ScienceDirect highlights the enormous void still present in the volume of carbon research and the volume of biodiversity research – there were 24,517 articles added in 2012 containing 'carbon emissions' to just 8,132 for 'biodiversity' (Figure 10). That said, recent large scale reports (MA, 2005; TEEB, 2010) have served to provide a platform for further research, and the formation of IPBES should help solidify this platform as the IPCC did for carbon.

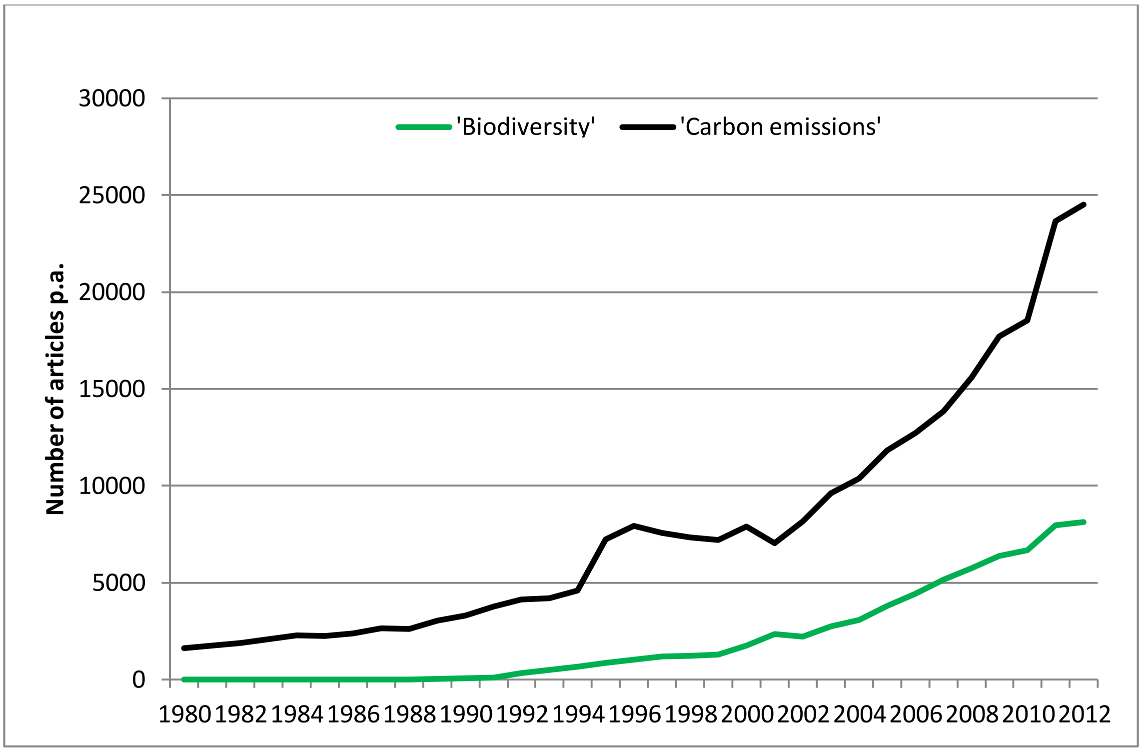

Figure 10: ScienceDirect results for 'carbon emissions' and 'biodiversity' search queries; Data: (ScienceDirect, 2012)

5.4 Voluntary offsets and the target of guilt

5.4.1 Ease of offsetting

The voluntary carbon market has been heavily criticised for its seeming lack of quality assurance (see 3.4.2.4), but this problem is very much being addressed by a number of verification standards. The purveying feeling in the general public is that carbon offsets are an easy fix, requiring a simple afforestation programme or development of a more efficient stove for developing nations. This may be a contributing factor to the explosion of OTC carbon offsetting companies. The irony is that even afforestation programmes are incredibly difficult to implement and can cause a wide array of problems; a widely used species for afforestation is *Eucalyptus.* There has been extensive research on the negative effect of *Eucalyptus* plantations on streamflow and water use efficiency of ecosystems (Hubbard, et al., 2010); ironic when the objective of the plantation is to alleviate the onset of climate change, likely to have an almost identical effect on water resources.

Biodiversity offsets developed for a wholly voluntary market are likely to be far more difficult to implement. As discussed at length, biodiversity and ecosystem functioning are two incredibly complex subjects, with vast gaps in our knowledge of them. As such, development of offset sites will require developers with specialist knowledge and experience of management for net gain of biodiversity. However, this may ensure quality assurance *per se*; small companies looking to hitch a lift on the emerging market and flood the market with cheap, low quality credits will be put off by the difficulty of offset development. Whilst encouraging, this may simultaneously have a negative effect on the future of biodiversity offsetting: one of the main advantages of the small voluntary carbon market is arguably its effect on public awareness and mainstreaming of the issue of carbon emissions (Kossoy & Guigon, 2012).

5.4.2 How easy to identify is a 'biodiversity impact'?

Carbon voluntary programmes are still vastly overshadowed by mandatory schemes in terms of volume and value (Kollmuss, et al., 2008), but one major achievement that can at least partly be attributed to the burgeoning voluntary carbon market is the raising of public awareness of the issue of carbon emissions and their specific targeting of consumer guilt; consumers are now looking for carbon neutral products, energy efficiency, and voluntarily offsetting travel.

One of the factors that has contributed to this is the 'accessibility' of carbon emissions, especially with regards to travel. Carbon emissions are accessible because of the direct link between transport and emissions; fossil fuels are burnt in order to power transport, and this emits CO_2. CO_2 is bad, and this impacts the environment. This is termed a carbon impact, or carbon footprint. The carbon markets have been able to capitalise on this to extend this basic understanding to emissions in supply chains and beyond. How would a voluntary biodiversity market categorise a 'biodiversity impact'?

This is an incredibly difficult question to answer; biodiversity impacts are being constantly assessed in EIAs and SEAs, but there are no likely candidates for a simple, 'accessible' biodiversity impact like the demonization of transport for carbon emissions. The most likely candidate is agriculture (McLaughlin & Mineau, 1995; Reidsma, et al., 2006), but this is supply chain dynamics, and is far removed from the end consumer; as a result, this is fairly intangible to the everyday consumer. Carbon emissions from transport are themselves an impact on global biodiversity through reasons highlighted in 5.2, but this is even more far removed and indirect.

The difficulty in defining biodiversity itself and the indirect nature of biodiversity impacts means that the positive effect of the voluntary carbon market is unlikely to be emulated in a similar biodiversity voluntary market.

5.5 Timing, timing, timing

Unlike carbon, which established itself during a period of prolonged economic growth, the current peaked interest in biodiversity market mechanisms comes during one of the worst economic recessions since the Great Depression of the 1930s (see Figure 9). Furthermore, with the threat of a double-dip recession, and the spread of the southern debt crisis in the eurozone, the economic prospects are looking grim for the near future. Despite this, Madsen and colleagues report that whilst there may not be “dramatic growth”, there is still “steady activity” in worldwide biodiversity markets (Madsen, et al., 2011, p. 27).

The immediate future is therefore uncertain for biodiversity markets – the global recession has cut off the main driver of biodiversity offsetting, public infrastructure projects and private development; whether biodiversity offsetting is likely to follow carbon’s trajectory depends largely on whether these activities resume pre-2008 levels. Furthermore, there are concerns that this arrest in the demand for credits may adversely affect the U.S. wetland market. 2010 saw the launch of 114 new wetland, stream and species banks, but many believe this to be more to do with the completion of pre-recession projects rather than a show of economic robustness (Madsen, et al., 2011).

5.6 Offset location

Biodiversity offsets, unlike the wealth of options afforded to installations wishing to cut their carbon emissions (3.5.5), are restricted by the guiding principles underlying the policy, especially like-for-like offsetting. Birdlife International emphasises this in their ‘key principles for biodiversity offsetting’: “there should be a presumption that offsets will be located as close as possible to the damage” (BirdLife International, 2010, p. 7). Current offsetting frameworks demonstrate this restrictiveness of offset location (Table 6).

Table 6: How offset policies in the US, EU, Australia, and Brazil address offset location; adapted from (McKenney & Kiesecker, 2010)

Offset policy	Offset location
US wetlands mitigation	Same watershed
US conservation banking	Same service area (US FWS); provides best long-term benefit to species
EU Natura 2000	Same biogeographic region in the same Member State; same bird migratory path
Australian native vegetation offsets	Adequate geographic link between losses and offsets; closer to on-site when losses are high significance
Brazilian industrial offsets	No preference, but if impacts are to a protected area, offset must benefit that protected area
Brazilian forest offsets	Same watershed

Unlike carbon offsetting, where there is a presumption that cutting emissions is not bound by location (see 3.5.1), the success of biodiversity offsetting depends directly on this geographic bounding. Whilst this preserves the integrity of the system, and prevents offsetting in general becoming a 'licence to trash', it can severely limit development in conservation bottlenecks. This can already be seen with Cumberland plain woodland in the Sydney basin, detailed in 4.2.2, where there is fierce competition between private landowners and developers.

The Sydney basin may prove to be an exception though; there has been increasing interest in landscape-level conservation planning, especially from conservation NGOs (Pressey & Bottrill, 2009; Kiesecker, et al., 2010). Wider landscape level conservation planning can help to identify regions that can support biodiversity offsetting, and earmark areas for potential offset sites. This requires a large amount of planning on an SEA scale, and this may serve to hinder the acceptance of biodiversity offsets.

5.7 The importance of additionality, or added value

Carbon emissions reductions under the CDM must be real, measurable, verifiable, and additional. Of these, the concept of additionality is considered as one of the most important aspects of a project. The general definition of additionality is outcomes that occur that are above and beyond outcomes that would have occurred under baseline scenarios (Schneider, 2009). In the context of afforestation and reforestation (A/F) projects under the CDM, a project is additional if:

"the actual net greenhouse gas removals by sinks are increased above the sum of the changes in carbon stocks in the carbon pools within the project boundary that would have occurred in the absence of the registered CDM afforestation or reforestation project activity" (eftec, IEEP, et. al, 2010, pp. 166-167)

Any methodology for determining additionality for a CDM project should follow the additionality tool provided by the UNFCCC, which comprises of four analyses:

- **Legal and Regulatory Additionality Test (Regulatory Surplus)**: a project must go beyond compliance to other official policies, regulations, or industry standards
- **Barriers Analysis**: if a project overcomes significant non-financial barriers (e.g. local resistance, institutional barriers, etc.) then it is additional
- **Investment Analysis**: revenue from providing credits must be a decisive reason for implementing the project (i.e. without sale of credits the project would be financially unviable)
- **Common Practice Analysis**: determines the extent to which other similar projects have been proposed or are being implemented in the geographic region

In terms of a comparison to biodiversity, restoration and creation projects (the preferred method of offsetting) are most similar to A/R programmes under the CDM. Biodiversity offset additionality is then defined as:

"A property of a biodiversity offset (or any action), where the conservation outcomes it delivers are demonstrably new and additional and would not have resulted without the offset" (eftec, IEEP, et. al, 2010, p. 5)

The concept of additionality when concerned with biodiversity offsetting is important, and has been highlighted by almost ever report assessing the utility of such a system (Treweek, et al., 2009; BirdLife International, 2010; eftec, IEEP, et. al, 2010; Comerford, et al., 2010; Madsen, et al., 2010; Madsen, et al., 2011). Furthermore, the idea is already enshrined in current offset systems. U.S. conservation guidance released in 2003 states: “land used to establish conservation banks must not be previously designated for conservation purposes (e.g., parks, green spaces, municipal watershed lands)” (USDOI, 2003), whilst NSW offset regulations require offsets to be “additional to actions or works carried out using public funds or to fulfil regulatory obligations” (NSW DECCW, 2005). With the current host of environmental legislation, especially within the EU, any biodiversity offset site must have gains above and beyond the current legislation (The Legal and Regulatory Additionality Test or Regulatory Surplus). Furthermore, despite increased environmental awareness and protection, the offset must also not have similar biodiversity projects within the same geographic area (Common Practice Analysis).

5.7.1 Additionality in Europe

The first thing to highlight about biodiversity additionality in Europe is the increased levels of land abandonment, especially in Eastern Europe (42% of all agricultural land in Latvia in 1990 was abandoned by 2000) after the collapse of socialism (Prishchepov, et al., 2012). As a result, additionality is unlikely to be achieved for scrubland and forest restoration/creation projects in these regions as these types of habitat are increasing regardless.

Furthermore, Member States also have considerable offset obligations for Natura 2000 sites under the Habitats and Birds Directives, mainly Article 6(4). Birdlife International has warned that Natura 2000 sites should categorically be outside the scope of any biodiversity offset programme for this reason (BirdLife International, 2010). Outside Natura 2000 areas, Member States have obligations towards habitats and species of Community Interest; Favourable Conservation Status (FCS) must be maintained for these species and habitats. Therefore any offset site must supply outcomes above and beyond FCS. Whilst this may sound difficult, FCS for one species may not be as favourable to other species, and the IEEP and eftec remain confident that there

“is likely to be scope for improving the ecological quality of habitats beyond those required to meet Favourable Conservation Status standards (as required in Natura sites under the Habitats Directive) and therefore benefits could be significant unless

additional national or site specific management measures are required and envisaged" (eftec, IEEP, et. al, 2010, p. 77)

In this way, so long as additionality is achieved, biodiversity offsetting in Europe on top of current environmental legislation could prove a significant conservation force.

5.7.2 Additionality elsewhere

The trend of land abandonment in Europe is not an isolated case, and global trends of land abandonment have been increasing since the 1700s (Figure 11).

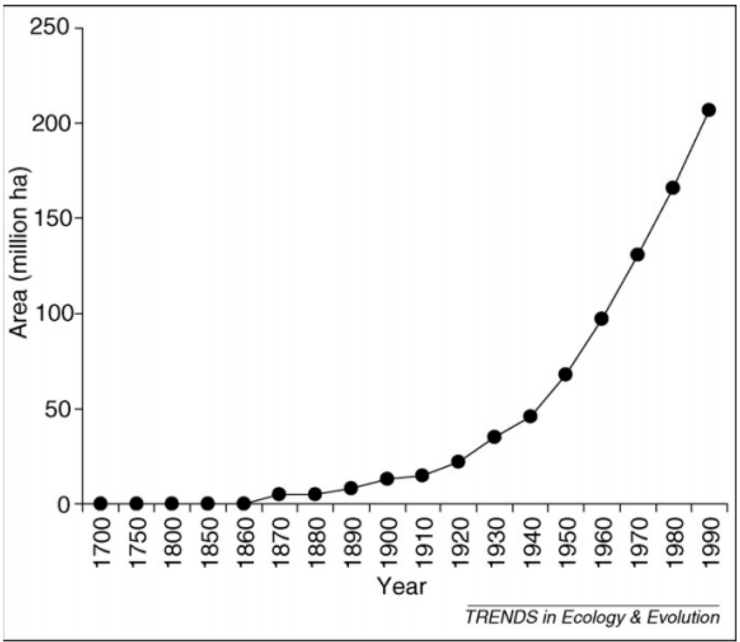

Figure 11: Abandonment of croplands from 1700-1990; Source: (Cramer, et al., 2008)

However, the environmental legislative force of the EU is globally rare, although there are promising trends in the increase in environmental protection legislation (e.g. Vietnam's 2008 Biodiversity Law, especially Decree No. 113). This means that passing the Regulatory Surplus test for additionality anywhere bar the EU and possibly North America will be significantly easier. This can be seen as a double-edged sword: on the one hand it paves the way for an explosion in biodiversity offset sites, but on the other hand it carries the risk of significant quality assurance issues outside of already heavily protected areas.

5.8 Building on current frameworks and policies

The substantial existing environmental framework in the eurozone has already been extensively covered; Member States have experience assessing and mitigating environmental impacts through the EIA and SEA Directives, as well as some experience with offsetting itself through the Habitats and Birds Directives and the Natura 2000 network. Although perhaps not as comprehensive as in the EU, EIA laws are beginning to be adopted worldwide and are currently present in South America (Brazil, Argentina and Chile), Asia (Japan, South Korea, China, Mongolia, Pakistan, Thailand, Malaysia, Russia and India), with even some in early stages in Africa (Madsen, et al., 2010). Whilst the existence of an EIA law is in no means a veritable precursor to a biodiversity offset system, it does suggest a raised environmental awareness, and willingness to establish a system where such a system could realistically develop.

One policy that has managed to embed itself firmly in the development of biodiversity offsets is the mitigation hierarchy (Figure 3); in a review of selected offset frameworks, McKenney and Kiesecker "find strong support for the mitigation hierarchy" (McKenney & Kiesecker, 2010, p. 167). The idea that offsets should be a very last resort after avoiding and minimising impacts is well entrenched, and after its development in the U.S. wetlands compensatory mitigation banking system, it has since been adopted in the EU and Australia. However, although there is worldwide adherence to the mitigation hierarchy, McKenney and Kiesecker point out that there is a paucity of quantitative guidelines for this decision-making process, and identify it as a key challenge facing the future of biodiversity offsetting (McKenney & Kiesecker, 2010).

One proposed system is a blend of landscape-level conservation planning and the application of the mitigation hierarchy, dubbed 'Development by Design' (Kiesecker, et al., 2010). Kiesecker and colleagues emphasise a shift away from the 'business as usual' scenario of project-by-project mitigation towards a framework more consistent with sustainable development. They argue that the proposed framework not only provides guidelines towards mitigation hierarchy decisions, but also provides a structure for conservation funding. However, the 'Development by Design' concept has attracted little specific interest.

Forest Trends' BBOP, however, has attracted an international collaboration of over 75 stakeholders; this includes a broad range including, among others, the European Bank for Reconstruction and Development (EBRD), Defra, Birdlife International, and the Rainforest Alliance. BBOP's objective is to engage even more stakeholders, and to implement further pilot projects in order to base decisions on more practical experience. However, the most crucial work undertaken by BBOP is progress towards an internationally agreed standard on biodiversity offsets that details, in a quantitative and indicator-driven way, how to approach adherence to the mitigation hierarchy (BBOP, 2012). The current standard will be updated based on field trials and a public consultation, before re-release sometime in 2014.

The work of BBOP is neatly complemented by the parallel work of UNEP-Wildlife Conservation Monitoring Centre (WCMC) and the Biodiversity Indicators Partnership (BIP) on the use of national biodiversity indicators and their development (Biodiversity Indicators Partnership, 2011). The BIP have developed a comprehensive framework that can be used to identify indicators for use in the application of the mitigation hierarchy (Figure 12).

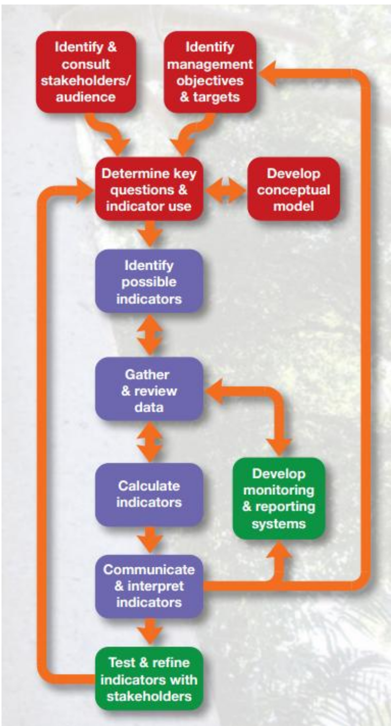

Figure 12: BIP's framework for developing biodiversity indicators; Source: (Biodiversity Indicators Partnership, 2011)

Whilst it is clear that sound policies and frameworks exist, there needs to be extensive capacity building. BBOP recognises this, and has conducted training with a range of stakeholders, including their training programme 'Biodiversity for Banks' (B4B), in association with the Equator Principles Association and the World Wildlife Fund. B4B aimed at aiding banks incorporate the true value of nature into their accounting. Even the EU, with its comprehensive biodiversity policies and legislative frameworks, will require a substantial increase in capacity; the DG Environment already has an incredibly high workload assessing damaging development projects, whilst many conservation agencies are already struggling keeping pace with existing legislation (especially in many eastern European countries where Natura 2000 sites have only recently been identified) (eftec, IEEP, et. al, 2010). Insufficient assessment and evaluation of mitigation measures in the UK has even led some to suggest the establishment of a new government body (e.g. an Office for Standards in the Environment), in order to meet these capacity requirements (Hill, 2009).

For an environmental market to succeed as far as the carbon market has today, it must have adequate measures for ensuring quality assurance, transparency, and monitoring and compliance. With the somewhat incomplete science that a biodiversity market would be based on, this requires a *substantial* framework. The US, with one of the most sophisticated environmental protection systems in the world, is struggling with the issue of compliance and monitoring of its two flagship schemes. That said, there are now certainly enough high profile biodiversity markets to learn from in the design of future systems.

5.9 Other considerations

5.9.1 Baselines

The definition of baselines is a crucial component in calculating credits in an environmental market, and there is no exception for biodiversity. Most often, the baseline is set as the 'business as usual' scenario, or what would have been lost if the project had not been started.

Broadly speaking, baselines could be defined in an identical manner to the baselines set under CDM methodology, ensuring that they are set:

- in a transparent and conservative manner
- on a project-specific basis
- accounting appropriately for national and/or sectoral policies

Compliance with the EIA and SEA Directives, as always, provides a strong starting point for determining baseline biodiversity scenarios. However, there have been criticisms that biodiversity baseline setting is inherently temporally biased; current baseline setting usually only takes into account 50 years prior, which for many components of an ecosystem (mainly trees) is very short. More scrutiny of the palaeoecological record could illuminate the distinction between 'normal' ecosystem functioning and unprecedented degradation (Willis, et al., 2005). Fortunately, a substantial baseline (including placement in a wider palaeoecological time period) has been provided by the comprehensive Millennium Ecosystem Assessment, which placed current extinction rates in a broader geological timeframe (see Figure 1) (MA, 2005).

Defining baselines on a project-by-project basis may have to be tuned slightly; as Keisecker and colleagues point out, the advantages of a landscape-level conservation plan are considerable when compared to business as usual project-specific planning (Kiesecker, et al., 2010). Unlike CDM projects, biodiversity projects benefit from greatly from a landscape approach. Attainment of Kentula's *landscape success* (Kentula, 2000), in this case the maintenance of biodiversity, depends heavily on conservation management at the landscape-level (e.g. minimum wetland densities and proximities detailed in 4.1.1.4).

Another pitfall to be avoided when setting biodiversity baselines is ensuring that regulations are stringent enough to avoid perverse incentives. The setting of baselines could theoretically penalise countries that have exemplary environmental records, and reward those with areas of higher degradation, as most conservation actions will be seen as 'additional'. This is not a problem specific to a biodiversity market; allegations were levelled at China after the Kyoto Protocol that Chinese installations had increased their outputs in order to be seen to be drastically cutting emissions.

5.9.2 Permanence

In order for biodiversity offsetting to become a viable conservation tool, offset sites must be managed in perpetuity, rather than covering the time period of project impacts. There may be opportunities for shorter term offsets, but the majority of current policy frameworks prefer management in perpetuity (McKenney & Kiesecker, 2010). Regulations stipulating long-term management ensure that a project continues to deliver.

A possible solution to this problem is to follow the framework of land use, land use change and forestry (LULUCF) projects under the CDM for addressing permanence issues, distinguishing *temporary* CERs (tCERs) and *long-term* CERs (lCERs). Whilst tCERs are issued for LULUCF A/R projects for a specific commitment period, they can be renewed or replaced if the project passes re-verification. Alternatively, lCERs are issued for the full length of LULUCF A/R projects, and only expire if the project fails re-verification or the project's crediting period ends.

Associated with the concept of permanence is the issue of offset timing. Common sense dictates that ideally offsets sites should be fully functional at the time of the impact if there is truly to be 'no net loss' (BirdLife International, 2010), and the majority of current policy frameworks agree (McKenney & Kiesecker, 2010). The two exceptions are the U.S. wetland mitigation system and offsets for Australian native vegetation, which provide a more flexible approach to timing. This is largely in response to the two significant barriers to implementing immediately functioning offset sites, namely:

i. **Foreseeability:** if offsets are required to be like-for-like, then this requires an unrealistic amount of forethought from developers as to the likely project impacts. This is especially true with ecosystems that take longer to create or restore.
ii. **Financial barriers:** if credits cannot be issued to a potential bank, it imposes significant upfront financial barriers to overcome before any returns can be made.

However, flexibility with implementation timing can be just as damaging to the integrity of a scheme than requiring immediate functionality is to developing banks; as already mentioned in 4.1.1.3, delays in implementing wetland projects may be contributing to a consistent net *loss* of U.S. wetland functionality (Bendor, 2009). Furthermore, the Environmental Law Institute (ELI) found that 90% of U.S. wetland banks sell some credits before achieving *any* performance standards (ELI, 2002). As a result, Moilanen and colleagues have called for the incorporation of uncertainty and time discounting into calculating offset ratios (Moilanen, et al., 2008). Bekessy and colleagues support this view, emphasising the risk of investing in potential future biodiversity gains against the extinction debt; the biodiversity bank must be a *savings* bank, and not a *lending* bank (Bekessy, et al., 2010).

Understanding the role of risk in the development of a biodiversity market is incredibly important in fostering third party involvement. Risk is already a major component of the carbon market, signalled by the higher price of brokered credits in regulated markets, and bilateral, OTC transactions in the voluntary markets (eftec, IEEP, et. al, 2010).

5.9.3 Leakage

The third and final consideration when assessing the future of biodiversity offsetting is the concept of leakage, which is now well established in the field of conservation (Oliveira, et al., 2007; Ewers & Rodrigues, 2008). It can be broadly split into:

i. **Activity shifting**: this is the direct displacement of an activity from within a project boundary to outside the project boundary
ii. **Market leakage**: e.g. if an area of land is no longer available for arable crops, the market price for these products increases, leading to land clearance elsewhere to take advantage of the high market prices

Research suggests that for A/R projects under the CDM (most comparable to creation/recreation ecosystem projects), leakage is only an issue on land with high opportunity costs (i.e. land that was originally used for producing commodities), and can be dismissed as negligible for land with low opportunity costs (eftec, IEEP, et. al, 2010). Whilst this means that the tracts of abandoned land in Europe and worldwide described in 5.7 are unlikely to experience much leakage, it has huge implications for the challenge of feeding nine billion people and the exponential increase in agricultural produce required (Godfray, et al., 2010). The agricultural market (especially in the EU) is a powerful market, and any developing biodiversity market will have to take into account competition with the agricultural market, and identify regions where this will be most intense. In this way, a biodiversity market partly depends on the development and trajectory of the local agricultural market.

5.10 A biodiversity disclosure project?

In section 3.6, it was established that a large factor for the success of the CDP can be attributed to the high cost of non-disclosure, especially for companies with a high level of public scrutiny (Verrecchia, 2001), and larger companies with more foreign sales (Stanny & Ely, 2008). The most important conclusion from this is that none of these factors are dependent on the properties of carbon; the costs of non-disclosure are not exclusive to carbon, and this can be seen in the rapid development of the CDP Water Disclosure Project (CDP WDP).

The CDP WDP was set up in 2010 and has quickly gained the backing of 354 institutional investors, representing assets of US$43 trillion. The first letter of request had a response rate of 60% (CDP, 2011); as Sarni points out in his book *Water Corporate Strategies*, voluntary reporting is no longer voluntary (Sarni, 2011). Water, like carbon, has now been identified as an important material risk to business and investors.

Meyer & Kirby assert that becoming a contemporary business leader revolves around internalising externalities, or the “societal problems that really can be laid at your doorstep” (Meyer & Kirby, 2010, p. 1), and one step towards this is full transparency and disclosure of your operations. In this way, it is no wonder that the CDP and CDP WDP have become such powerful informational forces so quickly; disclosure provides net gains for a business, according to Meyer & Kirby. There have been attempts at operationalising natural capital accounting for the internalising of externalities; one framework found that its methodology was generally applicable, and whilst finding that there were substantial benefits of the system with negligible costs, the system has not received much business interest (Jones, 2003).

So is there scope to create a biodiversity disclosure project (BDP)? As laid out in 1.2 and 5.2, biodiversity, although inherently linked to climate change, is a material risk in its own right. A report as early as 2004 by F&C Asset Management concluded exactly that: that biodiversity poses a tangible and material threat to businesses (Table 7) (F&C Asset Management plc, 2004). The report also highlighted nine high risk sectors.

Table 7: Material risks to companies presented by biodiversity; Source: (F&C Asset Management plc, 2004)

Risk	Example
Access to land	Access to new sites is affected by a company's track record on protecting/restoring biodiversity and water resources
Reputation	A biodiversity-related campaign over an issue such as Genetically Modified Organisms or dolphin-friendly tuna, reduces consumer confidence in a brand or company, resulting in lower sales
Access to capital	Environmental credit risk is assessed as high due to a company's poor biodiversity track record or management plans, and cost of capital increases
Access to markets	Inability to meet specifications from substantial buyers – such as government departments and agencies – for sustainably-sourced raw materials like timber, restricts access to a major market
Security of supply	Reduction in the quality and availability of essential materials such as fish
Relations with regulators	Concerns about a company's track record on biodiversity management, or lack of confidence in the quality of its biodiversity survey and management plans, leads to permit delays or fines
Liabilities	Unforeseen impacts of activities on biodiversity leads to financial liability even though a company's regulatory licences have not been exceeded

Despite this, Mulder & Koellner recently found that only 5 out of 50 banks studied had taken considerable steps to account for biodiversity, and that the majority of banks only perceived biodiversity as a 'reputational risk' (Mulder & Koellner, 2011).

Furthermore, a social and environmental reporting (SER) study showed that not even half of the FTSE100 companies investigated reported on all 20 of the components of biodiversity reporting proposed by Grabsch and colleagues (Grabsch, et al., 2010). The research concluded that "the overall level of biodiversity reporting is low" (Grabsch, et al., 2010, p. 12); this is echoed by the conclusions of Mulder & Koellner: "the general awareness of biodiversity being a business-relevant issue for the [banking] sector is at present quite low" (Mulder & Koellner, 2011, p. 118).

There is gathering momentum for businesses to disclose their externalities. The Corp2020 project (www.corp2020.com) has a similar vision to Meyer & Kirby; the project calls for a seismic shift in the corporate environment through their 'Planks of Change'. One of these 'Planks of Change' is disclosing externalities.

5.10.1 PUMA Environmental Profit & Loss (EPL)

Leading the way in this new age of transparency and disclosure is German multinational Puma SE, officially branded as PUMA. In 2011 they released the first ever Environmental Profit and Loss (EPL) statement, an attempt to monetise and internalise the environmental costs of their supply chain from production to stores (PUMA, 2010). Although the statement indicated environmental costs of EUR145 million, it was "far from the public-relations disaster that some had predicted" (Sukhdev, 2012, p. 27); the firm is now focussing on reducing the significant impacts (c.95% of overall impacts) of its manufacturing and raw materials supply chain.

Although the statement concentrated mainly on GHG emissions and water usage, it did attempt to internalise impacts on biodiversity and ecosystem services. "Loss of biodiversity and ecosystem services" was included and measured as the area of ecosystem converted for PUMA operations. Whilst this may seem simplistic, PUMA (rightly) points to the conclusions of the MA that land-use change is one of the principal factors driving biodiversity loss (MA, 2005). PUMA's land-use impacts contribute 26% to their overall impacts (Figure 13), largely as a result of their dependence on cotton, leather, and rubber.

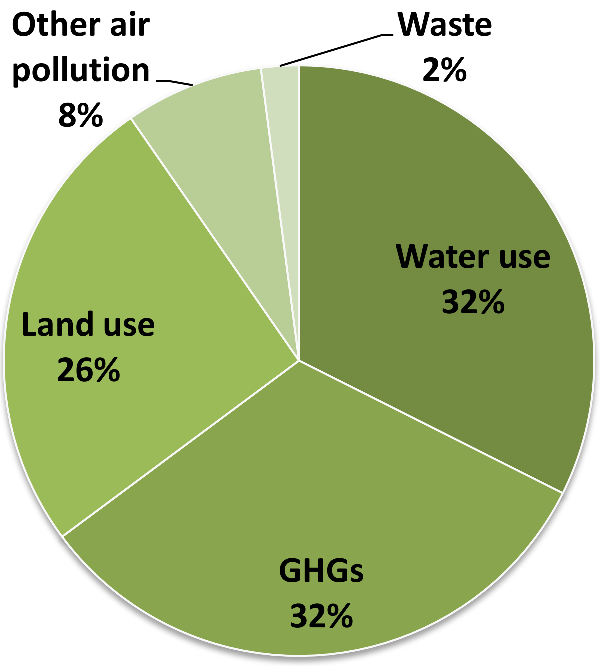

Figure 13: Distribution of PUMA's EUR145m environmental impacts; Data: (PUMA, 2010)

The per hectare impacts of PUMA's land-use impacts ranged from EUR63 for arid grasslands in Pakistan to EUR18,653 for coastal wetland in the US. Weighted averages then brought the overall average per hectare impact value to EUR347, indicating the majority of PUMA's operations are towards the lower value end.

PUMA's methodology followed extensive previous research, principally the TEEB study (TEEB, 2010), to calculate their impacts, indicating that there are already adequate methodologies to support a BDP. The disclosure itself was further aided by some strikingly honest built-in assumptions. Cattle-rearing is known for its enormous environmental impacts of incomparable water consumption, land-use change and natural GHG emissions. Despite some camps claiming that leather is a pure by-product of beef production, "PUMA takes the more conservative view that since the value of the hide adds up to 15% to the value of the cow, demand for leather forms part of the economic case for cattle-rearing and therefore part of the case for land conversion" (PUMA, 2010).

PUMA's wholly voluntary EPL statement has arisen for exactly the reasons that a BDP would be useful for; two of the benefits of the EPL that PUMA highlights are its use as a 'risk management tool' and as a 'transparency tool'. One of the crucial aspects of the CDP is identifying emerging risks and opportunities for stakeholders; PUMA states that the EPL "provides an early view of emerging risks, enabling us to respond strategically to protect and enhance shareholder value" (PUMA, 2010, p. 4).

5.10.2 Earthmind's Green Development Initiative (GDI)

After the unforeseen success of PUMA's first EPL, other major multinationals may follow suit, looking at their supply chain impacts. GDI's 'BioAreas Standard & Registry' is an attempt to aid in this pursuit: to create a common biodiversity unit, the BioHectare, based on area (similar to PUMA's EPL, but more complicated). The initiative is currently in its third phase, and evolved from Target 11 under Goal C of the CBD's Strategic Plan for Biodiversity: by 2020, 17% of terrestrial and inland water, and 10% of coastal and marine areas are effectively managed by, *inter alia*, "effective area-based conservation measures" (CBD, 2010).

The GDI defines a BioArea as:

"a geographically-defined area which is registered and managed to achieve specific conservation and responsible use objectives in the context of sustainable development" (GDI, 2012, p. 3)

The GDI, although in its early stage of development, sets out a rough management guide, including general criteria and indicators, and a guide to the process of registering as a BioArea (GDI, 2012):

- Candidate areas will be put forwards as Nominated BioAreas
- After a period of two years, the management of the BioArea must present an Action Plan, Baseline Assessment, and a Monitoring and Reporting Procedure.
- The BioArea will then become a Registered BioArea; GDI plans to maximise transparency by keeping a registry of all BioAreas at www.bioareas.org (under construction at time of writing)

- Registered BioAreas can also undergo independent third party certification to become a Certified BioArea (CBA); all CBAs will be clearly referenced on the registry

The GDI's BioAreas is certainly not the first initiative or attempt to define useful, applicable conservation areas; the idea is a crucial issue in modern conservation and the literature is somewhat saturated – most notably Myers and colleagues' famous biodiversity hotspots (Myers, et al., 2000), Olson and colleagues' delineation of the world's 867 ecoregions (Olson, et al., 2001), and Mittermeier and colleagues' identification of wilderness worldwide (Mittermeier, et al., 2003). However, scientific research into the subject tends to be over-technical and very much unrealistic; the GDI, if it gathers momentum, is simplistic in nature and could prove accessible to companies for biodiversity disclosure.

5.10.3 Fertile ground?

Whilst PUMA may be pioneering voluntary environmental disclosure, other industries are well accustomed to environmental accounting of sorts. Companies in the extractive industry, for example, are already used to a high level of biodiversity disclosure, as identified by Grabsch and colleagues (Grabsch, et al., 2010). Rio Tinto, who has had enormous environmental impacts in the past, are keen supporters of biodiversity offsetting as a conservation mechanism. This is somewhat unsurprising, as 38% of their operations are in proximity to land of "very high" biodiversity value (Rio Tinto, 2011). Shell is also very aware of the development of biodiversity in relation to business, co-authoring the report *Building Biodiversity Business* with IUCN (Bishop, et al., 2008).

The most crippling criticism levelled at biodiversity markets in general is its dependence on the incomplete science of biodiversity; a BDP could essentially circumvent this by relying solely on the simple measurement of area, as per PUMA's EPL and GDI's BioAreas. This coarse grain analysis may not be quantitative enough for financial markets, but may give a satisfactory indication of a business' biodiversity impacts.

As such, there seems to be fertile ground for the creation of a BDP: PUMA has voluntarily championed what a BDP statement might look like, factors contributing to the success of the CDP seem not to be exclusive to carbon, and the rapid success of the CDP WDP all appear to foreshadow a successful BDP. However, despite the existence of a fertile ground, a BDP does not look likely to be seeded in the near future; research like Mulder & Koellner's study suggests that there needs to be an exponential increase in the awareness of biodiversity and the material risk it presents *per se* before one is adopted.

CHAPTER SIX

6 CONCLUSIONS

"Our analysis shows that tradable permits are not a panacea for reconciling conflicts between economic development and conservation. Nevertheless, it would be wrong to conclude that they should be rejected. Their success depends on the existence of certain economic, institutional, and ecological preconditions" (Wissel & Wätzold, 2010)

The conclusion that Wissel & Wätzold draw from their research exhibits the typical irresolution seen when assessing the efficacy of market mechanisms for dealing with biodiversity loss. Research tends to focus on finding these "economic, institutional, and ecological preconditions" in some sort of perverse Goldilocks scenario. Unfortunately, biodiversity conservation does not have the luxury of time. Whilst compiling this research, another report has been released that adds to the growing concern at modern extinction rates: Collen and colleagues warn that a fifth of invertebrates globally could be at risk of extinction (Collen, et al., 2012). The precautionary principle has been touted by many as the saviour of the environment, but it may well be more of a double-edged sword, paralysing scientists and policy-makers in the process of decision making. Catastrophic levels of biodiversity loss need immediate attention and action: so can biodiversity offsetting follow in carbon's footprints and provide a step-change in attitudes towards biodiversity impacts?

6.1 Economic preconditions

There is undoubtedly a need for establishing a private market for biodiversity conservation in order to address an enormous shortfall in funding. As an example, the difference between the annual budget for the U.S. Department of Defence (US$525.4 billion for 2013) and that of the Environmental Protection Agency (US$8.3 billion). Similar trends are seen worldwide (e.g. the £1 billion shortfall in UK funding identified by (Lawton, et al., 2010)). There have been vast volumes of research attempting to decide how to distribute these limited resources (Wilson, et al., 2006; Bode, et al., 2008; Bottrill, et al., 2009; MacKenzie, 2009). A popular approach is the concept of ‘triage’, borrowed from medicine, where priority species are identified and dealt with over less vulnerable species (see (Bottrill, et al., 2008) for a full definition).

A large number of studies have identified the potential for a biodiversity offsetting scheme as a cost-effective means to address this financial shortfall (ten Kate, et al., 2004): for example, Comerford and colleagues find that there is potential for raising £53 million p.a. (Comerford, et al., 2010), whereas Kiesecker and colleagues found that in Wyoming, US$24.5 million was established as a mitigation fund for a *single* oil and gas field, compared to the US$4 million available to the Wyoming Wildlife and Nature Resource Trust p.a. (Kiesecker, et al., 2010).

In this way, a biodiversity offset market can be seen to have similar potential to the early stages of the carbon market for raising funds for biodiversity conservation. However, regardless of how a biodiversity market *could* alleviate conservation funding, it is dependent on the successful establishment of one, which in turn depends on more important institutional and ecological preconditions.

6.2 Institutional preconditions

Existing policy frameworks for biodiversity offsets have proved remarkably robust to the difficulties of establishing an offset scheme. The U.S. wetland compensatory mitigation banking system has been operating since the late 1970s, and conservation banking has quickly followed suit. The EU already has both a considerable existing environmental protection framework and substantial offset obligations already in place as a result of the Habitats and Birds Directives and the Natura 2000 network, and Australia has managed to successfully establish a number of regional schemes. The growing number of EIA laws worldwide suggests that institutional and policy frameworks capable of hosting successful offset schemes are already being developed.

One of the most challenging criticisms levelled at the existing policy frameworks is inadequate monitoring and compliance; this is seen to be rife by research into the U.S. wetlands system (Ambrose, 2000; Turner, et al., 2001). However, it appears that the ease of which compliance and monitoring can be attacked by critics of the system has evolved from the unrealistic expectations of a ‘silver bullet’ for global conservation efforts; a scheme that is perfectly designed and that halts the loss of biodiversity within a generation. No scheme, especially one as big as the U.S. wetlands mitigation banking, is perfect first time round – for example, the UK financial system, one of the largest and oldest worldwide, is still being exploited and facing considerable compliance issues (e.g. the recent Libor scandal and HSBC money laundering allegations).

As such, any large biodiversity market will be an iterative learning process (BirdLife International, 2010). Matthew & Endress point out the benefits to compliance of more appropriate objectives for mitigation banks (Matthews & Endress, 2008), and the incorporation of this type of research can aid the learning process. More importantly, the increased transparency afforded by the USACE RIBITS system in response to criticisms of opacity shows the surprising flexibility of a large institutional framework.

The establishment of a biodiversity market has similar institutional tools at its disposal that the carbon market benefited from; it has even been suggested the two can be linked through the creation of biodiversity and carbon credits via multiculture A/R projects (Bekessy & Wintle, 2008). Much of the success of the EU ETS can be attributed to the EU as an institution. Furthermore, the inclusion of market-based mechanisms for carbon emissions at Kyoto was largely a result of the success of SO_2 trading in the U.S. to combat aid rain; the U.S. already has two large biodiversity markets with which to display the potential benefits of such a scheme. Finally, the most infamous absentee in ratifying the Kyoto Protocol, the US, already has a significant vested interest in the development of a global biodiversity market, and would likely ratify a similar protocol on biodiversity.

6.3 Ecological preconditions

The crippling issue that a biodiversity offsetting market will never overcome is its dependence on the uncertain and incomplete science of biodiversity and ecosystem services. Regardless of how strong and robust the institutional framework, it cannot provide stability when balanced on such unstable foundations.

At the most basic level, there is just a critical paucity of data; this is best shown by the uncertainties still plaguing one of the forerunners in the arguments for protecting biodiversity: whilst the evidence is strong, the relationship between biodiversity and ecosystem services has yet to be categorically defined (Balvanera, et al., 2006; Hooper, et al., 2006). This is compounded by vast voids in biodiversity knowledge highlighted by Mora and colleagues in the latest global species estimate (Mora, et al., 2011); data deficiency forms a large part of a recent report warning of global invertebrate decline (Collen, et al., 2012).

Unlike carbon emissions, biodiversity is incredibly hard to quantify, something considered essential when creating an environmental 'currency' (Salzman & Ruhl, 2000). The course grain approach of equating biodiversity to a well managed area of land proposed by PUMA and the GDI, whilst attractive, ignores important ecological factors like edge effects, metapopulations, and most importantly, the extinction debt. Shape and area of a reserve has long been known to dictate its ecological success (Diamond, 1975), something likely to be overlooked when dividing parcels of land into easily quantifiable units.

Finally, regardless of the ecological success of restoring ecosystem functioning (which, as shown by the U.S. wetlands system in 4.1.1.3, is only moderately successful), there will inevitably be ecosystems that cannot be created or even restored. UK examples have been identified as glacial features: limestone pavements, turloughs, and pingoes (eftec, IEEP, et. al, 2010). There are potential concerns that a large biodiversity market will place a price on even these irreplaceable ecosystems, paving that way for further conservation shortcuts.

6.4 Conservation policy triage

With evidence for a global biodiversity crisis mounting, conservation policy-makers are faced with an ever more difficult decision. Whilst swift action is undoubtedly required, it is important to effectively prioritise conservation policies. For biodiversity, this is especially important when considering the current lack of public awareness; a BBC Radio Four survey completed in 2010 revealed that a large portion of the UK population thought biodiversity was "some kind of washing powder" (BBC, 2010) (this is accompanied by the tragic irony that 2010 had been designated the International Year of Biodiversity by the UN). This ignorance can even be seen to perforate the financial sector (Mulder & Koellner, 2011), despite the risks presented by biodiversity loss (F&C Asset Management plc, 2004).

Unfortunately, this public ignorance is somewhat mirrored in the scientific world. It seems that, like ground-level conservation itself, smart policy decision making will be based on the concept of triage: the efficient allocation of resources, financial or informational. In this case, the dependence on the establishment of a global biodiversity market comparable to the modern day carbon market should be categorised as beyond help; the qualitative nature of biodiversity itself fosters a lack of information and transparency that will prove, unlike carbon, to be irreconcilable with financial markets. Biodiversity offsets are designed to neutralise residual impacts, and it seems that the policy of biodiversity offsetting seems destined to provide a similarly residual contribution to more effective global conservation policies.

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