OXFORD

Nature in the Balance

The Economics of Biodiversity

Edited by DIETER HELM & CAMERON HEPBURN

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Dieter Helm and Cameron Hepburn

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AB	Agglomeration Bonus
AES	agri-environment schemes
ALBA	Bolivarian Alliance for the Peoples of Our America (Spanish: Alianza Bolivariana para los Pueblos de Nuestra América)
ANDP	adjusted net domestic product
BACI	Before–After–Control–Impact
BBS	Birds Breeding Bird Survey
CAP	Common Agricultural Policy of the European Union
CBA	cost–benefit analysis
CBD	Convention on Biological Diversity
CEA	cost-effectiveness analysis
CEH	Centre for Ecology and Hydrology
CGE	computable general equilibrium
CITES	Convention on International Trade in Endangered Species
CREP	Conservation Reserve Enhancement Program
CSERGE	Centre for Social and Economic Research on the Global Environment
DBH	diameter at breast height
Defra	Department for Environment, Food and Rural Affairs
DID	difference-in-differences
EKC	Environmental Kuznets Curve
ELS	Entry Level Stewardship (ELS)
ES	Environmental Stewardship
ESA	US Endangered Species Act
FAO	Food and Agriculture Organization
FGM	Farm Gross Margin
FoE	Friends of the Earth
GAMS	Generalized Algebraic Modeling System
GBIF	Global Biodiversity Information Facility
GDP	gross domestic product
GEF	Global Environment Facility
GF	Go with the Flow
GHG	greenhouse gas
GIS	geographic information system

GM	genetically modified
GNI	gross national income
GPL	Green and Pleasant Land
HUC	hydrologic unit code
IBAT	Integrated Biodiversity Assessment Tool
IC	incremental costs
ICDP	integrated conservation and development projects
IIA	independence of irrelevant alternatives
IIED	International Institute for Environment and Development
INBio	National Biodiversity Institute
InVEST	Integrated Valuation of Ecosystem Services and Tradeoffs
ITQs	Individual Transferable Quotas
IUCN	International Union for Conservation of Nature
LEFT	Local Ecological Footprinting Tool
LETS	Local Exchange Trading Systems
LMMC	Like-Minded Megadiverse Countries
LS	Local Stewardship
LSOA	lower super output areas
LULC	land-use and land-cover
MA	Millennium Ecosystem Assessment
MENE	Monitor of the Engagement with the Natural Environment
MN-GAP	Minnesota Gap Analysis Project
MPCA	Minnesota Pollution Control Agency
MVP	minimum viable population
NCBG	Nash Cooperative Bargaining Game
NDP	net domestic product
NEA	National Ecosystem Assessment
NEMA	National Environment Management Authority
NGOs	non-governmental organizations
NLCD	National Land Cover Database
NPV	net present value
NS	National Security
NW	Nature at Work
OECD	Organisation for Economic Co-operation and Development
PA	protected area
PES	payments for ecosystem services
PIC	prior informed consent

PPP	purchasing power parity
PVNB	net present value
REDD +	Reducing Emissions from Deforestation and Degradation
RFF	Resources for the Future
SNA	System of National Accounts
SPM	site-prediction model
SSSIs	sites of special scientific interest
TEEB	The Economics of Ecosystems and Biodiversity
TGF	trip-generation function
TSI	trophic state index
UK NEA	United Kingdom National Ecosystem Assessment
UKCIP	United Kingdom Climate Impacts Programme
UNCED	United Nations Conference on Environment and Development
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
WCED	World Commission on Environment and Development
WHO	World Health Organization
WM	World Markets
WTP	willingness to pay
WWF	Worldwide Fund for Nature

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1

Introduction

Dieter Helm and Cameron Hepburn

The scale of biodiversity destruction around the world is widely acknowledged. Nature is 'in the balance'. If the current trends are extrapolated, then the scale of the current extinction will be as great as, or greater than, the five mass extinctions that have occurred on Earth over the past 540 million years. But this time, the mass extinction will not have been caused by a meteorite or some exogenous factor, but rather by humans. And because we are the cause, it is not inevitable. The challenge is to come up with credible agreements and policies to first reverse and then restore.

To date, such efforts have been largely ineffective. Part of this is the sheer complexity of the problem. It is not like climate change. Biodiversity is highly heterogeneous (there are many relevant species), it varies from geography to geography, and even fundamental summary statistics are still rather uncertain. In contrast, greenhouse gases are globally well-mixed, there are relatively few of them, and their concentrations in the atmosphere are known. Indeed, as we shall see, the problems are confounded by the fact that the concept of biodiversity itself is poorly understood and poorly defined. Furthermore, there are no easy aggregate indicators, such as global average temperature, to simplify the analysis, and the eventual damages from biodiversity loss are potentially even more uncertain than the damages from climate change—and of course the two problems interact. The features that both problems do share—such as having significant impacts that are in the very distant future—do not encourage politicians to impose the costs of mitigation now.

In order to begin to formulate policies which might have some chance of first limiting the decline of biodiversity and then reversing it, the concepts need to be defined. This is a task for scientists and economists: a joint endeavour. Following on from the overview chapter by Dieter Helm and Cameron Hepburn, in Part I: Concepts and Measurement, Georgina M. Mace's *Biodiversity: Its Meanings, Roles, and Status* (Chapter 3) provides a scientist's perspective. She notes that 'biodiversity' is just the compression of two words (biological diversity). But whilst diversity is a familiar concept for economists to get a handle on, the difficult part is to define the domain—diversity of what? This is almost always going to be context-dependent. But such complexity does not mean that progress towards designing appropriate policies is necessarily inhibited, or impossible. On the contrary, what it implies is that there can be few, if any, generic policies. Whilst nations can agree a Convention on Biological Diversity (CBD, established in 1992) and Convention on International Trade in Endangered Species (CITES), the implementation has to be local or, at the most aggregate, regional. Context is inescapable—and so therefore is the measurement and mapping.

Kathy J. Willis et al.'s *Identifying and Mapping Biodiversity: Where Can We Damage?* (Chapter 4) sets this measurement and mapping problem in the economic context. Their chapter starts with the assumption that there is going to be more damage to biodiversity. Increasing global pressure on land for food, fuel, industrialization, and population growth makes this inevitable. The immediate task is, therefore, to work out what biodiversity should be protected, and which might be sacrificed. In addressing this stark reality, Willis et al. take us through the informational requirements, compare these with what we have already in place, and look at the tools already available. They divide the information into several categories—and in the process illuminate the complexity. These categories include: richness, vulnerability, fragmentation, connectivity, resilience, and the impact on future biodiversity. In the process, they identify not only what we know, but also the gaps in the knowledge base, and the sorts of tools that we need to develop to make rapid and effective assessments when projects come forward which will damage biodiversity.

The sorts of exercises that Willis et al. have in mind can be conducted at a variety of geographical levels. One such is to take stock of a country's total biodiversity, or some component of that biodiversity. In the UK, there has been a comprehensive attempt to evaluate the national ecosystem services (one dimension of biodiversity). In Chapter 5, Ian J. Bateman and Grischa Perino et al.'s *The UK National Ecosystem Assessment: Valuing Changes in Ecosystem Services* provides a summary of the National Ecosystem Assessment (NEA) exercise which they led. The purpose of the exercise is to value changes in different scenarios. They describe these scenarios as: 'go with the flow', 'green and pleasant land', 'local stewardship', 'national security', 'nature at work', and 'world markets'. For each of these scenarios, the value of ecosystem services provided is calculated against a baseline. These then have to be defined and measured. The NEA is the bringing together of a series of alternative possible future scenarios and the valuation of the ecosystem services. The authors do not claim precision—this is not the purpose. Rather it indicates the relationship between the value of the services and the choices made at a national level, and hence—as with the Willis et al. chapter—brings the detail of measurement to bear on the general objectives of limiting harm.

Whereas Part I focuses on the contribution of scientists and the importance of science to economic valuation, Part II focuses on economic techniques, and in particular valuation and cost–benefit analysis (CBA). These approaches have always been controversial, and indeed many economists have not helped by making exaggerated claims about pricing nature. CBA is a particular set of techniques, which cast partial and limited light on the allocation of scarce resources in project appraisal. The techniques employed have largely been from the toolkit of microeconomics, and have focused on the marginal impacts of projects that are too small to impact on the economy as a whole.

In Chapter 6, Giles Atkinson et al.'s *Valuing Ecosystem Services and Biodiversity* provides a guide to the principal techniques and the current state of play in what is inevitably a technical territory. But they go further, and build on the ecosystems approach used in the NEA. The treatment of non-use costs and benefits, the value transfers from one context to another, and the problem of aggregating over ecosystems are just some of the areas of CBA which need to be taken into account in utilizing this sort of approach when formulating policy.

A partial step from the marginal to the aggregate was represented by The Economics of Ecosystems and Biodiversity (TEEB) exercise. At the broader political level, once the Stern Review (2007) had revealed how powerful a global aggregate estimate of costs and benefits of mitigating climate change could be in shaping the debate (whatever the merits of the analysis), there was a push to do the same thing for biodiversity. Yet for many of the reasons stated, not least the conceptual difficulties in defining the concepts and the context-specific nature of impacts, any such exercise was probably doomed from the outset. The TEEB exercise is the largest-scale attempt so far, but it clearly has not yet had the impact of the Stern Review, and has necessarily been limited by the heterogeneous and context-specific nature of biodiversity. Chapter 7 by Pavan Sukhdev et al., *The Economics of Ecosystems and Biodiversity (TEEB): Challenges and Responses*, addresses a set of concerns about the application of economics in this domain, including concerns that economic valuation is inherently subjective, ecological values are incommensurable, incorporating economics just makes a hard problem even harder, and that marketization of nature involves attempting to harness the very same powerful forces that have been responsible for environmental degradation to date. The ultimate challenge to the TEEB exercise—that it did not live up to expectations—is met with the implicit response that those expectations were never realistic in the first place.

The measurement of ecosystem services focuses on the flows of services provided. An alternative line of attack is to start with assets—and in particular natural capital. In Part III, Edward B. Barbier's *Natural Capital* (Chapter 8) provides an overview of the concept of natural capital, and Kirk Hamilton's *Biodiversity and National Accounting* (Chapter 9) shows how the concept has been incorporated into accounting at the national and international levels. Barbier links the concept of natural capital to ecosystems. Once nature is thought of as capital, there is an obvious link to the idea of sustainability and sustainable development by setting out the conditions necessary to maintain the value of that capital through time. Barbier sets out the implications for both weak and strong sustainability criteria in terms of the degree of substitution between different types of natural capital and between natural and other types of capital. Hamilton picks up this theme, building on recent World Bank work, notably *The Changing Wealth of Nations* (World Bank, 2011). Natural accounts need balance sheets, and Hamilton sets out the potential treatments of biodiversity in the national balance sheet, and how net income and net savings can be measured, before providing a number of empirical estimates.

Biodiversity is both local and global, and tends to be concentrated in 'hot spots' such as rainforests. With much biodiversity already destroyed in developed countries, there is a particular need to focus on developing countries. In Part IV these international development issues are explored. In Chapter 10, Charles Palmer and Salvatore Di Falco's *Biodiversity, Poverty, and Development: A Review* looks at the relationship between biodiversity, poverty, and development. They examine relationships between biodiversity, ecosystem services, and economic development. They observe that development pressures (perhaps created by population growth) can lead to environmental degradation and biodiversity loss, intensifying poverty, and increasing pressure on the household to meet its subsistence needs, in turn leading to exacerbated degradation and biodiversity losses. Even in those limited cases where actions to protect biodiversity (such as protected areas and bioprospecting—the search for valuable compounds from wild organisms) are effective, they often fail to benefit the rural poor in developing countries. The trade-off between biodiversity protection and economic development can be very real, which simply exacerbates the policy challenge.

Recognizing the global dimensions is one necessary aspect of making progress on limiting the damage. But so far credible and effective international agreements have been noticeable by their absence. In *Regulating Global Biodiversity: What is the Problem?* (Chapter 11), Tim Swanson and Ben Groom use a standard game theory approach to the problem, indentifying the classic features of free-riding and the distribution of costs and benefits to show why the CBD has been of limited impact so far. Their focus is on the nature of the bargains—and why, until this is explicitly addressed, further progress is unlikely to materialize soon.

Finally, Part V turns to policy instruments and incentives. The obvious point of departure is to consider whether existing biodiversity policies work—and in particular which ones have done better, and which have made little difference. In Chapter 12, Daniela Miteva et al.'s *Do Biodiversity Policies Work? The Case for Conservation Evaluation 2.0* concentrates on the three policies most commonly used in developing countries: protected areas, payments for ecosystem services, and decentralization of natural resource management. They set out the appropriate conservation-evaluation techniques. The limited number of rigorous conservation-evaluation studies suggests that while protected areas can in some cases be moderately effective, all three approaches leave much to be desired.

In Chapter 13, Stephen Polasky et al.'s *Are Investments to Promote Biodiversity Conservation and Ecosystem Services Aligned?* focuses on the alignment between the objectives of enhancing ecosystem services (to humans) and conserving biodiversity for its intrinsic value. While the two objectives are related, they may in theory conflict. Increases in biodiversity need not necessarily increase ecosystem services to humans, and vice versa. However, payments for ecosystem services can support greater biodiversity where, instrumentally, biodiversity enhances ecosystem services that improve human well-being. From a study in Minnesota, Polasky et al. observe that, in practice, the difference between the two objectives is not too large—targeting one of the objectives generates 47–70 per cent of the maximum score of the other.

A neglected aspect of policy design is the impact of ownership on incentives. In Chapter 14, Nick Hanley et al.'s *Incentives, Private Ownership, and Biodiversity Conservation* fills this gap with an evaluation of a number of land incentive schemes and, in particular, agri-environmental schemes policies. They compare and contrast approaches based on regulation, uniform payment schemes, conservation auctions, conservation easements, and the creation of markets for biodiversity. They show that when contracting with private landowners, what matters is the detail of policy design—in particular the precise allocation mechanism, the specification of contract outputs, the duration of the contract, and the price.

A particular policy problem arises when a species is threatened with extinction. In the final chapter, Jo Burgess et al.'s *On the Potential for Speculation to Threaten Biodiversity Loss* considers what happens as species become rarer and hence supplies from the wild dwindle and prices increase. They show that extinction may be an incentive-driven process, especially where there are stockpiles of the associated wildlife commodities such as ivory, tiger skins, and so on. The policy implications are very much empirically dependent on the particular cases, but attention is directed to the existence of these private incentives, and may call for public intervention—but not always, especially if public agencies can commit to strict conservation.

These chapters add up to much more than the individual contributions, important though they are. Together they provide a comprehensive assessment of the state of the economics of biodiversity and what economic analysis has to bring to the design and implementation of measures to protect and enhance biodiversity. The link between the economics and the science runs right through the book, in understanding the heterogeneity of the concepts of biodiversity and the thresholds and limits pertaining to renewable and non-renewable resources.

So far, economics has had little impact on policy design and it is no accident that this neglect of economic valuation, economic accounting, and the use of prices and incentives has been paralleled by the neglect of biodiversity. The current situation is not good, and the consequences of business-as-usual will be a global experiment every bit as worrying as the related and parallel unfolding of climate change. A mass extinction episode over the coming century, building on the destruction of recent decades, warrants not only serious analysis, but also the development of better policies. Treated as separate from the economy, and without regard to the economics of biodiversity, the destruction will go on largely unchecked. Economics is a necessary tool, grounded on good science, and cannot be ignored if progress is to be made.

2

The Economic Analysis of Biodiversity

*Dieter Helm and Cameron Hepburn**

2.1 INTRODUCTION

Biodiversity loss should be regarded as one of the greatest economic problems of this century for two reasons. First, it is economic growth and development that has caused biodiversity loss and ecosystem degradation.[1] The rapid expansion of the population from around 2 billion in 1900 to over 7 billion today, combined with the enormous growth in income and consumption, have already wreaked havoc on our planet's natural ecosystems. Current environmental pressures will only increase as the human population swells from 7 billion to 9–10 billion by 2050 (UN, 2011), and as the number of so-called 'middle class' consumers grows from 1 billion to 4 billion people (Kharas, 2010), driven by a materials-intensive growth model (Baptist and Hepburn, 2013).[2] This economic growth has led to dramatic reductions in poverty but also severe ecosystem degradation.[3]

Second, future losses of biodiversity and ecosystems may significantly reduce the productivity of our economic systems. By 2100, on current projections, we may have eliminated half the species on Earth (Wilson, 1994;

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[1] Ecosystem degradation and biodiversity loss often, but not always, accompany one another. For an assessment of the degree of alignment between policies to protect biodiversity and those to support ecosystem services, see Chapter 13 by Polasky et al. It concludes that: '[i]n general, investing in conservation to increase the value of ecosystem services is also beneficial for biodiversity conservation, and vice versa'.

[2] Middle class consumers are defined as those with daily per-capita spending of between $10 and $100 in purchasing power parity terms.

[3] Palmer and Di Falco explore the relationship between the two in Chapter 10.

Thomas et al., 2004), and raised global temperatures by more than 3.5 degrees centigrade (IPCC, 2007). The rainforests may have been largely deforested by then, the oceans depleted, and land degradation may have significantly affected agricultural productivity. We are living through one of the great extinction episodes in geological history.[4]

Technical progress might help to counter some of these negative effects. New techniques of efficiently manufacturing food, and new sources of energy, may facilitate a transition to 'green growth' and development. But this is far from certain. Massive biodiversity loss and climate change represent an unprecedented and enormous experiment with life on Earth, and it is astonishing that biodiversity is not a topic routinely covered in every standard economics textbook. Instead, one of the greatest resource allocation questions has been largely ignored by the mainstream economics profession. Dasgupta (2008) is correct to note that '[n]ature has been ill-served by 20th century economics'.

One of the reasons biodiversity has been relegated to the margins of economics is that there are formidable obstacles in the way of high-quality economic analysis. Biodiversity is a particularly intractable economic problem. It has system properties that defy easy definition. It is more than the aggregate sum of species: some species play a vital role in the survival of ecosystems; some provide key ecosystem services to humans; some are positively harmful to humans; species depend on each other; and policies aimed at biodiversity are often oblique, aimed at preserving habitats rather than particular species. Biodiversity is a series of overlapping public goods from the local to the global scale.

Conceptually, biodiversity shares a number of core issues with climate change—such as intergenerational equity—but is much more analytically demanding. Climate change is simple by comparison: the atmosphere comprises a set of gases, changes in its composition can be measured, and empirical estimates can be made of the relation between these changes in composition and temperature changes. From the science, a well-defined economics, policy, and political agenda have developed, focusing on carbon prices and related policy instruments. The additional difficulties facing the economics, policy, and politics of biodiversity are enormous, and can be observed from a comparison of impact of the Stern Review on Climate Change on the one hand, and the TEEB process on the other (see Sukhdev et al., Chapter 7).

Without a precise and operational definition, biodiversity is conceptually elusive. The economic tools are not particularly fit for purpose, though they are a helpful starting point, and in any event are all we have to work with. This

[4] For comparison with previous extinction episodes, see Barnosky et al. (2011).

matters not just for the purposes of analytical neatness: without a clear definition of biodiversity, it is hard to measure its loss, and harder still to design the appropriate policy instruments and evaluate the impact of such policies. For some, biodiversity loss is proxied by the number of species or populations; for others biodiversity policy should focus on preserving rainforests, wilderness, and specific areas as nature reserves and protected habitats.[5]

The structure of this chapter is as follows. Section 2.2 addresses the concept of biodiversity, its measurement, and the nature of the resource allocation problem. Section 2.3 reviews the standard valuation techniques that make up cost–benefit analysis (CBA). Section 2.4 briefly considers the substitutability of man-made and natural capital, depletion, and renewable and non-renewable resources. Section 2.5 considers the policy implications: the role for payments for ecosystem services (PES), eco-credits, compensation mechanisms, and the use of prices for the environment, in addition to the specification of protected areas. Section 2.6 considers the institutional dimension, notably the problem of biodiversity treaties and agreements. Section 2.7 looks at implementation and accounting, and the embedding of biodiversity within the core of economic policy. Finally, section 2.8 concludes.

2.2 WHAT IS BIODIVERSITY?

Given that governments and international organizations regularly produce biodiversity statements, agreements, and policies, it might be concluded that biodiversity as a concept is both well-defined and measured. It would then be possible to assess policies to see whether they increase or decrease biodiversity. But cursory examination leads to a very different conclusion: there is no obvious and agreed definition of biodiversity, and in practice there are a number of sub-definitions and concepts upon which policy is based. For example, according to the Food and Agriculture Organization (FAO), biodiversity encompasses the

> variety and variability of plants, animals and micro-organisms, at the genetic, species and ecosystem level, that are necessary to sustain the key functions of the agro-ecosystem, including its structure and processes for, and in support of, food production and food security. (FAO, 1999)

[5] See for example Weitzman (1998), who argues that the species located further apart on the phylogenetic tree are more valuable than those that are closer genetically to other species since the former possess more unique genes. See also Brock and Xepapadeas (2003) for another approach to valuing an individual species. For a presentation of a broader ecosystem approach, see Secretariat of the Convention on Biological Diversity (2004). Polasky et al. in Chapter 13 also consider the relationship between ecosystem conservation and biodiversity protection.

This defines biodiversity partly in terms of its contribution to economic production. Other definitions focus simply on the ecological aspects of the system.[6] In contrast, the Convention on Biological Diversity (CBD) defines biodiversity broadly 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, Article 2)

Any empirical estimate of the 'amount' of biodiversity is going to be a crude approximation at best. The concept is hugely informationally demanding.

Biodiversity is a shortening of the phrase 'biological diversity'. Biodiversity therefore has two components. The word 'biological' relates to life and living organisms, sometimes roughly captured by the notion of 'nature'.[7] The word 'diversity', meaning variety, can often be captured by an index. As Mace (Chapter 3) notes, biodiversity is therefore 'variation among units of life'.

Which units of life? We might consider the number of species, or the number of genes therein, estimate populations of specific species, or focus on even broader levels of communities, biomes, and ecosystems. Mace notes that it is arguable that genetic variation should be considered the fundamental unit for assessing biodiversity, as it represents the raw material for 'structure, form and function'. However, species often serve as natural units for the measurement of biodiversity, because species are largely well-defined units on which evolutionary pressures act,[8] and for the practical reason that lists of species are relatively straightforward to compile. But even here there are challenges, both scientific and economic. For instance, some species are clearly more important than others. The loss of tigers at the peak of a food chain might have a very different effect from the loss of a less charismatic species further down which might support an entire ecosystem—a 'keystone species'. Yet predators also can play key roles—without them herbivores flourish, changing the vegetation (Lotka, 1920; Volterra, 1931). Furthermore, from an economic point of view, some species are more highly valued by humans than others, both for their intrinsic value and for the services they yield. Thus corn, wheat, and sugar cane are highly valued, to the extent that other species are pushed aside, whilst mosquitoes are not. Attempts to economically define an 'optimal number of species' are therefore inherently difficult, if not impossible, and in practice much of the literature is confined to looking at marginal changes from the status quo.[9] So using species as the unit of life is not without

[6] World Resources Institute et al. (1992) and Noss (1990).

[7] Nature can also be understood to incorporate reference to non-living things.

[8] We can estimate critical thresholds beyond which a species is condemned to extinction, at which point it effectively becomes a non-renewable resource.

[9] It is not always clear what constitutes a marginal change. See Fisher et al. (2009).

its problems; Mace describes alternative units and their advantages and disadvantages in Chapter 3.

How should 'diversity' of these units of life be measured? One might seek a single, composite metric that captures all possible variation and measures of interest. But despite many attempts to develop such a metric, as Mace points out, this is doomed to fail. A single metric will always confound different attributes and inevitably lose interesting and important detail. At the other extreme, one might seek to compile a comprehensive suite of metrics to measure different facets of diversity. This approach, however, quickly becomes too complicated.

As a result, scientists commonly employ simple metrics, such as the number of known species in a particular location, as a proxy for diversity. The results are not dramatically different from concepts commonly understood to economists—diversity is a familiar concept in economic theory, and is measured in numerous economic contexts, notably in financial theory, regulatory economics, and in measures of energy security. In the biodiversity context, the relationship between the area of a habitat and its species density can be estimated.[10] Thus the claim that rainforests are biodiverse might mean that they have more species per hectare than other habitats. Indices to measure diversity along these lines include the Shannon index[11] and the Simpson index, which sums the squares of the proportion of each species in a given area. So, as with the Herfindahl index in industrial economics, a measure of zero represents infinite diversity (perfect competition), while a measure of unity represents no diversity (monopoly). More sophisticated biodiversity measures might apply ecological weights to the species. Furthermore, we might identify key indicator species—species occupying key niches in an ecosystem, and highlight them in a diversity measure. Another measure, following the FAO definition, might work by applying weights based on the economic productivity of the species. And so on. However, these simple metrics also have their obvious inadequacies. Species lists, for instance, are unlikely to be comprehensive, as probably less than 10 per cent of all the species on Earth have been described and named. In short, there is no perfect solution.

As if defining the relevant unit of life and agreeing a means to measure variety of those units were not hard enough, a further difficulty relates to the absence of a 'baseline' against which to evaluate current biodiversity. There is no real 'wilderness' left, and no 'balance of nature'. Humans have been modifying 'nature' since the Pleistocene, eliminating the mega fauna and

10 A 'species–area relationship' is often approximated by a power function of the form $s = cA^z$, in which s is the number of species, A is the area, and c and z are fitted constants (Preston, 1962).

11 The Shannon index quantifies the uncertainty in predicting the identity of a species when drawing individual units from a random sample of the total population.

hence changing the balance between forest and open plains. 'Nature' itself is best viewed as subject to continuous change, not a series of equilibria (Rohde, 2006).

The important point here is that what we measure depends on what we define as biodiversity, and that in turn depends on what the question is that the measure is supposed to answer. Economists typically start the analysis from a human perspective, placing values on individual species, and then aggregate upwards. This aggregation ignores the ecosystem properties, and this in turn means that aggregating individual valuations of species tells us little about the value of biodiversity as a whole. We return to this point later when we consider The Economics of Ecosystems and Biodiversity (TEEB) and related exercises in larger-scale system valuations.

Having explored the challenges in defining biodiversity, the next issue is to explore what sort of economic problems are raised by biodiversity. The economic approach to biodiversity sees the problem as one of market failure and, in particular, of externalities and public goods.[12] A public good is both non-excludable and non-rival, while positive externalities from biodiversity might potentially relate to both rival and non-rival goods. In both cases, there will be over-exploitation and under-provision. Without intervention, biodiversity has little value captured in a price and it will be under-provided by the private sector. Biodiversity shares this with climate change abatement, but climate change relates to a single global atmosphere, whereas biodiversity is a plethora of overlapping public goods. Public goods include species at one level, and national parks at another, right up to the Amazon and Antarctica. Ecosystems can also be described as public goods. These public goods may even conflict: preserving one species may reduce the availability of another. Preserving a national park may eliminate species that depend on human activities prohibited within national parks. The best economists can do is to identify which public goods are being pursued, and at what scale (from local to global), and then determine how best to design policy instruments appropriate for those public goods.

A particularly difficult feature of biodiversity is the critical thresholds: above a critical threshold, a species might be classified as a renewable public good; below it becomes non-renewable and condemned to extinction. These thresholds are uncertain, and hence in making decisions under uncertainty, the question of risk aversion rises. In much of the environmental literature, the precautionary principle—that we should be risk-averse in the face of such uncertainty—is evoked in this context, both at a species level and also more generally.

In sum, biodiversity is difficult to pin down conceptually, and there are various competing definitions that might be employed for quantitative

[12] See, for example, Fisher et al. (2009) for details and discussion on the classification of ecosystem services.

research. A further problem is that there are likely to be many species that have not yet been discovered, particularly insects and amphibians. While difficult, these problems do not prevent useful economic research, using tools and concepts including public goods, externalities, non-linearities and threshold effects, and the economics of information. One economic tool that is critical to biodiversity policy is the various economic valuation methods that have been devised, refined, and incorporated into non-market CBA.

2.3 VALUING SPECIES AND LANDSCAPES

Notwithstanding these conceptual difficulties, in order to address the problem that the market will under-provide biodiversity, and to incorporate biodiversity into markets, either property rights have to be assigned so that markets reveal prices, or techniques are required to elicit the economic value of biodiversity. CBA provides the current economic toolkit for valuation.

Methods to determine the valuation of a species are reasonably well developed in the economics literature, and there has been substantial progress over the last decade (see Chapter 6 by Atkinson et al.). Economists tackle this valuation problem in two ways: bottom-up, using the traditional tools of valuation of marginal changes in biodiversity; and top-down, where systems analysis is employed to attempt to generate an aggregate valuation. The Stern Review on the Economics of Climate Change provided a model of how valuation estimates could capture the political debate (Stern, 2007). The TEEB exercise, which some regarded as a Stern-type analysis, did not produce an aggregate value on biodiversity for fairly obvious analytical reasons. Given that biodiversity (and indeed the climate) are what might be described as necessary conditions for human existence, putting a precise number on the aggregate value of either is open to ridicule. It is a category mistake.

Economic valuation techniques try to place a monetary value on species so that they can be included in resource allocation and their conservation can be traded off against other uses of scarce resources. These 'valuations' are attempts to work out how much should be spent on conserving a species or habitat, given that the monies could be spent on some other—competing—end. They are not estimates of fundamental ethical value or of ecological importance, and confusing the two gives rise to a serious misunderstanding of economic valuation techniques. The question they address is a very limited one.[13]

[13] However, see Sagoff (2004), who provides an alternative view, claiming in effect that the ethical and the economic are conflated.

Valuation techniques are required because most species do not have markets, and hence market prices. Outside agriculture and zoos, they are not typically owned, and rarely traded. There are exceptions—for example, rare animals and plants are subject to collection, and collectors will tend to pay more the rarer they become. Indeed this relation of valuation to scarcity has its own dynamic: if a species (or some attribute of a species) can be stockpiled, then there may even be incentives to make it rarer. As Burgess et al. demonstrate in Chapter 15, stockpiling rhino horn and tiger parts may be a profitable strategy if the rhinos and tigers then are pushed towards extinction. A further example is the collapse of the Great Auk population in the North Atlantic through harvesting and hunting. This led trophy hunters to try to kill the last one. (They succeeded off Iceland in 1844.)

Economic valuation methods fall into one of three categories: revealed-preferences methods (including hedonic pricing and the travel cost method); stated preferences (including contingent valuation—carefully constructed surveys—and choice experiments); and production function approaches. Given the technical and informational problems, these techniques are best regarded as snapshots from different angles, each trying to approximate the underlying value, and in practice it is useful to compare and contrast the estimates.

Revealed-preference methods take a behavioural approach, trying to use the choices people make to reveal their underlying preferences by making clever use of econometric methods. For instance, the price we pay for assets like houses near particular habitats might carry a value premium that can be estimated, and the value of time and effort people expend to visit habitats in order to see particular species can also be calculated.[14] These valuation techniques are necessarily imperfect. All sorts of other factors affect house prices and our journeys. Standard controls can eliminate some, but not all, of this bias. But stated-preference approaches are vulnerable to biases too—in the way information is provided to the subjects, and in the way subjects respond. The practitioner is necessarily forced to choose the least worst method in the specific context, and where possible to compare the different snapshots that result. For this reason, economic valuation is very much a case study exercise.

In many cases, economic methods produce monetary valuations for species, biodiversity, and ecosystems that are considered ‘too low’ by ‘experts’, because they do not and cannot capture all of the relevant benefits of the species (though we may find out that a species has some deadly characteristics in the future). For many ecologists and environmentalists, the problems are so great as to render the techniques at best useless and at worst positively misleading.

[14] Thousands, for example, travel to Loch Garten in Scotland to see the ospreys.

For some, taking a human-only perspective is to take too narrow a view of nature: many argue that nature has intrinsic value.

These critiques conflate two different issues. Economic valuations are necessarily incomplete, but incompleteness is not a reason for discarding them. Rather it suggests that the role of the experts is to help to address the question more precisely. In any event, there are many circumstances where incompleteness does not matter. Consider the policy issue of whether to destroy some natural asset in order to build a road. Assuming the cost of building the road is known, all that matters is to discover whether even an incomplete economic valuation of the natural asset exceeds the costs. Where it does, then the decision to preserve the asset is determined. Where it does not, then more detailed analysis may be required.

Intrinsic value is a different issue. To claim something has value above and beyond human consideration raises a host of questions about where such additional value comes from, how it could be justified, and in particular what the objective basis upon which it must rely is. In designing biodiversity policy, such ethical considerations cannot of course be ruled out, but it is beholden upon those who advance this view to explain what should and should not be conserved, and to explain why resources should be expended in their preferred way, rather than on other alternatives. A number of green philosophical approaches take a cavalier view of the consequences of strong sustainability, and severe restrictions imposed upon permissible trade-offs can be made, but fail to explain how in practice the implications may work out.[15]

It might be argued that the difficulties are so great that CBA should be ruled out. But then, what are the alternatives, given that decisions have to be made? Resources are not infinite: preserving a species may mean that houses cannot be built or funds cannot be spent on something else. Resources are unfortunately scarce, and allocations have to be made.

To date, valuation has played relatively little part in biodiversity policy although, as we shall see, this may be changing. Decisions about species and habitat are typically administered through the application of rules and command-and-control regulation. Planning law gives the job of weighing up the case for and against a development that might harm biodiversity to an official or a judge. The answer tends to be binary: it either is, or is not preserved. Too often the implicit result is that the environmental considerations are assigned a value of zero.

This matters in two important ways. First, a lot of biodiversity is destroyed. Too little conservation takes place. Second, where the environment is damaged because there is no value assigned, there is no compensation. Compensation forgone means resources unavailable for conservation more generally. Without

[15] For a sample of views, see parts 4 and 5 in Dryzek and Schlosberg (1998).

a compensation mechanism, there are few opportunities to use economic incentives, such as payments for ecosystem services and eco-credits, to benefit species and habitat conservation. CBA provides a route to both better policy-making and compensation. It is highly imperfect, but it is hard to think of a superior alternative, and not employing CBA has often led to costly mistakes.

CBA forces costs and benefits to be made explicit: qualitative expert evidence tends to be more amenable to use in lobbying and implicit influencing. It is also important to bear in mind that decisions by experts assume that experts are independent guardians of the public interest. But experts have careers, interests, and views of their own, and can be lobbied and influenced. Experts can be hired not just by those who seek to protect biodiversity, but by those who seek to damage it too. Typically, developers have deeper pockets, and hence are better able to muster 'expert evidence' in their favour.

Thus, despite all the caveats, economic valuation and CBA provides an important tool for the design of biodiversity policy. It is one way of characterizing biodiversity problems, and because of the problems of assigning values to non-market goods and services, the assumptions always need to be spelt out. This includes the information basis and the consequences of changes in information. Unfortunately in many cases, monetary valuations are stated without the caveats, especially by politicians and those with interests in the outcomes.

2.4 DEPLETION, SUBSTITUTION, AND RENEWABLE AND NON-RENEWABLE RESOURCES

The optimal amount of biodiversity is not, as some environmentalists claim, that level which provides a 'state of nature' that existed before the human race evolved. As already noted, humans have been exploiting other species and depleting natural resources for their entire history, and now there is no true wilderness left. Indeed, it is not clear that a concept of 'pristine nature'—nature without humans—is in any sense optimal, although much of the ecology and conservation literature takes nature without humans as its baseline (and hence assumes it to be optimal).[16] The question is not whether to deplete natural resources, but by how much (see Chapter 4 by Willis et al.).

Two areas of economics are relevant here: the optimal rules for depleting renewable and non-renewable resources (biodiversity resources are primarily renewable resources but have some non-renewable features); and the substitutability of natural and man-made capital. By defining rules for the use of

[16] Willis et al. (2007), and Willis and Birks (2006).

natural resources, the concept of a sustainable growth rate can be formalized to meet the constraint that welfare must not fall in any future period (Pezzey, 1992).[17] Other definitions of sustainability impose the constraint that aggregate natural capital must not fall in any period, or even more specifically that the stock of biodiversity assets is non-decreasing for all future periods. This is where the concept of natural capital comes in (Barbier, 2011).

Renewable resources are those that can restock themselves, provided that resource abstraction is limited and managed. Fish stocks are a classic example. If humans harvest a limited amount of cod each year, say, the species can maintain a constant population. This might be seen as humans replicating nature (or indeed a recognition that humans form part of nature) in that predators reduce populations of prey, but some sort of balance is maintained between them in semi-stable systems.[18] But if humans harvest cod without limit, the population may collapse. The experience of perhaps the greatest cod fishery in the world—the Grand Banks—is an example of the latter.[19] Worse, as noted earlier, once a species is on a path to extinction, its value may rise, and this in turn provides incentives for even more rapid depletion—a vicious circle explored by Burgess et al. in Chapter 15.

The optimal harvesting rate for a renewable resource is one that ensures that the rate of return from investing in other assets, the market interest rate, is equal to the rate of return from the renewable asset. This often leads to the prescription that, after some initial adjustment, stock levels should be kept constant so that the harvest rate matches the natural growth rate of the stock. The optimal depletion rate for a non-renewable resource rests on the same concept of arbitrage, but generates a very different conclusion. The resource is not going to last, so the issue is not whether to deplete, but how quickly, and thus by implication, which set of people should have the benefit. Hotelling (1931) identified that, under specific assumptions, the optimal extraction path implies that the price of the natural resource increases at the interest rate. These assumptions are reasonably strong, and Livernois (2009) shows that the empirical evidence does not provide overwhelming support for the (simple) rule; modifications (such as better accounting for technological progress in extraction costs) are needed.

An important assumption is that non-renewable resources can be swapped for other physical or financial assets. But how far can man-made capital substitute for natural capital before the returns on man-made capital start to

[17] See Heal (2012).

[18] The concept of 'the balance of nature' is a useful heuristic, but it has limited empirical support, since change is a permanent feature. Even in 'predator–prey' models, the empirical support for the classic Lotka–Volterra equations in ecology is weak. From an economist's perspective this is unfortunate since the concept of equilibrium and modelling shocks is one that is familiar in economic theory.

[19] See Duncan et al. (2011) for background and modelling of the dynamics of the collapse.

decline? Consider a standard production function, which translates a series of inputs into output. Conventionally, neoclassical economics has two factor inputs, capital and labour. Capital is further disaggregated into human and non-human. In classical economics, there were three factors: land, labour, and capital. Land was subsumed under capital in standard models. However, economic theory in the last few decades has incorporated resources and/or natural capital as an additional factor of production,[20] of which biodiversity is one example. Biodiversity is then an asset which yields a stream of (eco) services.

The way the factor inputs are separated out reflects differing views of the relationship between humans and the natural world. One—classical—view is that if natural capital is considered to be a factor of production, the natural environment constrains the possibilities of humans, and once humans move up against the constraints, a Malthusian-style feedback comes into play. Humans can expand only so far before we run out of land, water, oil and gas, and so on. As biodiversity declines, these constraints become even tighter. Substitution of man-made capital for natural capital is feasible up to a point, but at a critical threshold we damage nature's ability to renew itself. The critical point can be termed the 'carrying capacity' of Earth, and in this view, it implies that the potential growth in the human population is ultimately limited, even if these limits may be in the more or less distant future.

Standard neoclassical models present a rather different view. Output growth is driven by technological progress, which improves the factor inputs of capital (human and non-human). We get better and better at making things. To facilitate this technical progress, we use up natural capital, but end up with much more non-natural capital: cities, infrastructures, goods, and services. We end up with fewer swallows, but more iPads. Constraints remain for non-renewable resources, but even here there is considerable optimism built into the standard neoclassical view. We can increasingly modify genetic material, creating new plants and animals as a result. It is not inconceivable that species could be recreated and that biodiversity, whilst altered, may be improved. Sequencing the human genome, and the developments in the biosciences, offer up new opportunities, and move the focus of biodiversity from the species to the genes themselves. In the plant world, it is possible to store seeds for very long periods, and one measure to protect biodiversity has been to create new seed depositories.

The Hartwick–Solow rule formalizes this view and the idea of a growth path based on the substitution between natural and man-made capital. It states that if the rents derived from the efficient extraction of a non-renewable resource are invested entirely in reproducible physical and human capital, and if there

[20] See Stiglitz (1974), Barbier (2011), and Hepburn and Bowen (2012).

is a high degree of substitutability, or a sufficiently fast rate of technological progress, then non-declining, sustainable consumption through time is feasible. This relates to the concept of weak sustainability and the feasibility of such a condition depends very much on the substitution possibilities open to an economy.

2.5 POLICY INSTRUMENTS

Armed with the economic concepts of externalities and public goods, what policy approaches are available for maintaining biodiversity to maximize social welfare? Economics offers three broad approaches. First, we can employ 'economic instruments' to create incentives to correct biodiversity-related market failures and to ensure optimal provision of biodiversity-related public goods and resources. Economic instruments can be divided into price instruments (such as taxes on damaging behaviour, or subsidies for biodiversity provision), or quantity instruments (such as tradeable permits), or some combination of the two. Second, we can regulate through 'command-and-control' (such as the specification of protected areas). Third, we can 'do nothing' on the assumption that the costs of intervention (government failure) are likely to be greater than the costs of the market failure. The third approach is the default, and biodiversity has been suffering.

2.5.1 Biodiversity-related externalities

The market underprovides biodiversity because there are positive externalities—the benefits are not entirely captured by the actor providing biodiversity. Similarly, the market overprovides goods that damage biodiversity because there are negative externalities—the full costs of pollution, fertilizer, pesticides, land conversion, and so on are not borne by the relevant actor. If something is to be done about these biodiversity-related externalities, the choice is between economic instruments or command-and-control.

The choice of approach for biodiversity externalities is related to a much more fundamental topic of economic thought. Debates between the merits of market and planned economies occurred at length in the 1930s. Claims of socialist planners (such as Oskar Lange) were pitted against market advocates (such as Friedrich Hayek). The general superiority of markets, compared with central planning, arises because of the ways in which incentives and information are economized in markets, compared with the computation demands placed on planners. In a market-based economy, individuals and firms make decentralized decisions based on the vector of prices, which emerge from the many decentralized decisions. In contrast, the planner needs to know all the production and utility functions in order to optimize. Incentives differ too: in markets, individuals and firms pursue utility and profit maximization; in planning, the social welfare function has to be derived from individual utility preference orderings, and the bureaucratic incentives to seek out rents need to be taken into account.

It is for these fundamental reasons that economists often start by looking for 'economic instruments' that correct prices and take advantage of markets, before turning to command-and-control. For biodiversity policy, however, the use of economic instruments and markets turns out to be very challenging. In fact, for understandable reasons, most policy is command-and-control, as Miteva et al. point out in Chapter 12. This is not to suggest that 'economic instruments' do not have potential, and recent contributions to the policy literature have suggested that more enhanced roles for markets are worth considering.

As already noted, there are two broad categories of 'economic instrument'. The first focuses directly on the prices faced by agents who are degrading biodiversity. A direct price can be established by taxing activities that cause biodiversity loss, or by establishing subsidy payments for ecosystem services (PES). Subsidizing ecosystem services is conceptually distinct from subsidizing biodiversity. Nevertheless, Polasky et al. in Chapter 13, in their study of conservation funding in Minnesota, US, find a high degree of alignment between strategies that target the value of ecosystem services and those that target habitat for biodiversity conservation. The appropriate price level has to be estimated, using the valuation techniques discussed earlier. For instance, the policy-maker might identify an externality (say the damage to bees caused by pesticides and herbicides), conduct a monetary valuation study, and then either impose a tax (in this example on pesticides and herbicides) or grant a subsidy for the under-produced service (in this case, beekeeping). In some cases, the instrument is applied to the cause of the biodiversity loss; in others, it is directed at the consequences. Causes of biodiversity loss are varied, but include agricultural chemicals, conversion and development of land that had supported wildlife, river pollutants, and waste products. Providers of biodiversity can also be supported by a range of measures, including an array of subsidies, conservation auctions, and conservation easements, which provide economic incentives for landowners to conserve biodiversity (see the so-called 'set-aside' policy, which provides land free from cultivation at field margins), and can support biodiversity policy, even if the underlying motive may be primarily to influence agricultural output. Other environmental schemes can subsidize particular farming practices that encourage biodiversity (Natural England, 2009).

A second way of creating appropriate economic incentives is to create new markets. This involves directly fixing the quantity of the externality, and allowing the market to determine the price. For instance, the quantity of pesticide could be fixed, and those wishing to use this chemical would have to apply for (or buy) permits for pesticide use, which could then be traded. The market price for permits is the level at which demand for permits (from agents) equals the supply (set by government). Where the damage is great, the chemicals might be banned, but in many cases, the optimal quantity of pollution is not zero. A variant of this approach is to require developers to purchase eco-credits to offset the impact of their development on biodiversity. Specific harms can be identified, and their values estimated through valuation techniques. Developers are then required to purchase eco-credits, generated from biodiversity-protection activities, of the same valuation before the development can proceed.

Such economic instruments have several potential drawbacks compared with 'command-and-control'. First, generalized economic incentives or trading schemes may result in problematic hotspots (Stavins, 2003). This is because biodiversity tends to be highly location-specific, and because the impact of policies to protect biodiversity (e.g. deforestation policies) is also likely to be a function of location. Furthermore, incentives to deforest land and destroy biodiversity vary dramatically from one location to another (see Pfaff and Robalino, 2012). Location-specific pricing and/or regulation may be required. Unlike climate change, where it does not much matter where carbon dioxide is emitted, spatial location matters enormously to biodiversity and ecosystems. Trading between ecosystems can be a recipe for disaster. However, command-and-control regulation does not completely avoid the location-specific issues either: it simply places these quantity choices (and therefore the implied price) in the hands of a regulator, who has to devise estimates of the location-specific costs and benefits. Location-specific direct regulations may also be preferred because otherwise trading volumes would be too slim for a market to work.[21] Nitrates, for instance, might be banned from use in certain sensitive environments, but be subject to taxes everywhere else. Second, as discussed later, economic instruments alone may not work for biodiversity protection when the biodiversity in question is a public good.

The choice between fixing the price and fixing the quantities—taxes and subsidies versus permits—depends on a wide range of factors. One factor often cited is efficiency under uncertainty, which depends on how rapidly the marginal benefits and marginal costs of the activity change as more of that activity occurs (Weitzman, 1974; Hepburn, 2006). Under uncertainty, the choice of instrument, loosely speaking, turns upon what we are most worried about. If it is critical that a certain threshold number of a particular species is preserved, then quantity rather than price instruments is likely to be more

[21] An interesting case study here is that of the proposals to price the abstraction and use of water (Environment Agency and Ofwat, 2011).

efficient under uncertainty. In contrast, if the costs of protection could skyrocket if a particular target turns out to be too stringent, it may be more efficient to fix the price. However, there are a whole host of other factors that need to be considered, not least the administrative feasibility and the politics of the different instruments (Hepburn, 2006). For instance, a major drawback of market-based mechanisms compared with taxes or command-and-control is that constructing artificial markets can involve developing a complicated set of market institutions. Governments may find this much more difficult to manage, especially in developing countries, than, say, simply creating and enforcing a protected area, or sending out the tax collectors.

So, empirically, which instruments work best for biodiversity? Unfortunately, Miteva et al. find in Chapter 12 that the evidence is too weak to draw clear conclusions. There remains a dire need to evaluate properly the different performance of biodiversity conservation approaches—there simply is not credible empirical evidence of what works and where. Miteva et al. find that protected areas do consistently stimulate modest changes in land use that may positively affect biodiversity. Despite billions invested in protecting ecosystems and biodiversity, however, the evidence base for economic instruments and other interventions is simply too shallow to say anything meaningful about whether they are superior to protected areas. Until economic instruments and market-based policies are tested in a manner that allows their subsequent evaluation, it will remain difficult to identify general rules about optimal biodiversity policy.

2.5.2 Biodiversity-related public goods

So far we have considered policy for protecting biodiversity through the lens of market externalities, focusing on correcting or creating markets, or regulation to require appropriate action of market actors. But, in many cases, characteristics of biodiversity suggest that it can also be viewed as a public good. For instance, the 'existence value' of biodiversity is non-rival—one individual's enjoyment of the existence of a species does not affect another's enjoyment of the existence of the species. Goods with such characteristics provide additional problems for the use of market-based instruments. For instance, even if a competitive market could be constructed, because the marginal costs of provision are zero, in a competitive market the marginal price would also be zero. A market with a price of zero obviously does not create any incentive to invest in the provision of the good.

Much biodiversity has multiple public-good characteristics. This constrains the policy instrument choice. For instance, the Amazon rainforest, the Snowdonia National Park, and sites of special scientific interest (SSSIs) are not obviously amenable to a simple-minded application of economic instruments for externalities, whether taxes, subsidies, or permits (though these instruments may help, and may indirectly provide a source of funding). There are two main options for public-good provision: the state provides the public good for free and recovers the fixed and sunk costs through general taxation; or the public good is turned into a club good, by giving some entity a monopoly right, and the legal power to exclude non-members.

Public ownership plays a key role through national parks and preserved areas on government-owned land. What happens within such parks tends to be command-and-control, though in principle the owner can create incentives, and economy-wide incentives may cover the domain within which the park is located. Non-government charitable institutions, such as large-scale environmental organizations, can also provide these public goods. In the UK, the Royal Society for the Protection of Birds (RSPB) has over one million members whose subscriptions fund the 'club' and whose reserves are sometimes made open to the public, but are usually for members only. The National Trust has 3.8m members, has a mix of open and restricted access, and similarly uses a membership fee. Wildlife Trusts are more local in their areas, but again use a mix of funding mechanisms. Together, these non-governmental institutions own a significant amount of land.

Many of these considerations apply in a context in which property rights and the rule of law are generally part of the institutional architecture. In contrast, in developing countries—where much biodiversity is concentrated—the circumstances are less amenable to the use of market instruments. But again resorting to command-and-control does not necessarily solve the resource-allocation problems and, in practice, what matters is the empirical evidence in particular circumstances, as Miteva et al. emphasize in Chapter 12. This is an area where more research is urgently needed.

2.6 TREATIES, TARGETS, INTERNATIONAL AGREEMENTS, AND INSTITUTIONS

Many biodiversity problems are international, in one of two senses. First, they may be transboundary, such that the causes and solutions involve at least two countries. Migrating species often cross national boundaries, and certain habitats provide biodiversity of potential or actual use to populations beyond national boundaries. The great African migrations of large herbivores are iconic examples, focused not just in the Serengeti, but also the Okavango Delta. Fencing, notably in respect of the Okavango, has major implications for these species. Bird migrations do not respect national boundaries, and the open seas are beyond national jurisdictions.

Second, some biodiversity problems are global, in that the ecosystem concerned provides a global public good with effects on humanity everywhere on Earth. Biodiversity that provides global public goods tends to be concentrated in so-called hotspots. The tropical rainforests are disproportionately important in terms of species densities. Some ecosystems, such as the Amazon, are genuinely global in significance—their services affect the entire climate of Earth. Even smaller, less significant ecosystems within national boundaries may have international significance if they appear in people's preferences. The international feature of biodiversity potentially makes valuation exercises more complicated, since the domain of preferences and the number of people potentially included is very wide.

International public goods introduce a number of issues: the design of international treaties, the bargaining between nation states, the relation between climate change initiatives and biodiversity, the potential trade-offs and connections between poverty reduction and biodiversity, and the design of institutions. Several major international treaties and initiatives relate to biodiversity. The CBD, which entered into force in 1993, is the most significant international agreement on biodiversity (see Chapter 11 by Swanson and Groom). It establishes that developed nations provide financial resources to support developing nations to meet the incremental costs of protecting biodiversity as required under the Convention. But the financial flows, channelled through the Global Environment Facility, are tiny compared with the value of the natural resources at stake. Swanson and Groom consider these payments in the context of a bargain, or game, between rich and poor countries. They note that the rich tend to offer to pay the poor the 'incremental costs' of protecting biodiversity. But this is an extreme negotiation outcome, in which all the economic surplus is captured by the rich world, and none by the poor. Swanson and Groom argue that this outcome cannot serve as an equilibrium based on narrow national self-interest (a Nash equilibrium), and identify conditions in which both threatened and actual destruction of biological resources by developing countries would be expected to be observed and, indeed, is observed. In short, the CBD is unlikely to provide adequate protection of global natural infrastructure because financial flows are too low and the underpinning concept of incremental cost is not a Nash equilibrium—even if the informational and enforcement problems could be overcome.

The CBD is complemented by other international agreements. The Convention on International Trade in Endangered Species (CITES) was drawn up in 1973, coming into effect in 1975, and was designed to limit international trade in wild animals and plants threatened with extinction. Other international agreements also impact indirectly on biodiversity. Of these, those whose primary focus is climate change are perhaps the most important. The Reducing Emissions from Deforestation and Degradation (REDD +) scheme under the UN Framework Convention on Climate Change (UNFCCC) is the main vehicle, which was also the focus of discussions relating to the CBD in Nagoya, Japan, in 2010. REDD + relates to biodiversity in two ways: reducing emissions limits climate change, which in turn protects biodiversity; and protecting key habitats, such as rainforests, typically (but not always) protects biodiversity while limiting emissions.

Considering these international dimensions as public goods provides a basis for considering the extent to which international agreements and treaties may 'solve' biodiversity problems. Scott Barrett categorizes global public goods according to: whether they can be provided unilaterally or by a small group of countries; where they depend on the weakest link; or where they depend on the combined efforts of all states (Barrett, 2007). This classification is helpful in the biodiversity case. Preserving the American bison can be solved by the US, and the snow goose depends on Canada plus the US. Some migratory species depend on the weakest link: for example, rare breeding birds in the UK are vulnerable to key countries on their migratory routes. Preserving the Amazon rainforest relies on a number of countries, and is so big as to demand global cooperation to preserve it. Thus, although there is a good case for an overarching international biodiversity framework agreement, different levels of international cooperation are needed for specific cases. Again biodiversity turns out to be much more complicated than climate change.

Biodiversity and climate change are examples of international problems, requiring international solutions. But they arise in a context of many competing international issues and priorities. Other policy goals may conflict with the protection of biodiversity. Indeed, this is why biodiversity tends to suffer when rapid development takes place. Rivers get polluted, forests cut down, air quality declines, and agriculture takes in more land. The trade-off between biodiversity and poverty and between biodiversity policy and poverty is explored by Palmer and Di Falco in Chapter 10.

The nexus between biodiversity and poverty creates particular challenges. Many of the areas rich in biodiversity are in poor countries. Economic growth is associated with the destruction of biodiversity, but so, too, are poverty traps—pressures (perhaps created by population growth) can lead to environmental degradation and biodiversity loss, intensifying poverty, and increasing pressure on the household to meet its subsistence needs, leading to exacerbated degradation and biodiversity losses.

Global and regional agreements require monitoring and enforcement, and supporting research capabilities. Coming to an international agreement depends on the creation and sustaining of credible institutions. In a number of global public goods cases this has been recognized with mixed success. The UNFCCC provided the institutional framework within which the Kyoto Protocol was established. The World Health Organization (WHO) binds those who agree to specific regulations. The diversity and fragmented nature of biodiversity problems makes an overarching institutional framework both hard to construct and difficult to sustain, as was demonstrated at the Rio + 20 conference in June 2012. The result has been to focus on regional mechanisms, which will reflect Scott Barrett's classification of public good problems noted previously.

Less recognized have been the implications of the application of the theories of bureaucracies and government failure to these institutional examples. Non-governmental organizations (NGOs) pursue limited objectives, and have membership to maximize. It is a notable feature of the biodiversity NGOs that they tend to specialize in one aspect. In the UK, the RSPB looks after birds; Plantlife looks after plants; Buglife looks after insects; and the National Trust focuses on landscapes (and buildings). Within the domain of these environmental groups, there are many conflicts and disputes, and often they fail to cooperate to exploit opportunities for biodiversity in general. This is all the more surprising given that their memberships overlap considerably. Then there are campaign groups—such as WWF, the Sierra Club, Friends of the Earth (FoE), and Greenpeace. Campaign groups campaign, and this requires specific rather than general objectives. Indeed, it is interesting to note that for FoE and Greenpeace it is their anti-nuclear activities that gain the most attention.[22] International conferences—such as Durban on Climate Change and Rio + 20 on biodiversity—are important recruiting opportunities for NGOs, which are provided with extensive media coverage.

The result of this institutional fragmentation has been that the political impact of the environmental movement has become less than the sum of its parts. When compared with other large-scale membership organizations, such as trade unions, the contrast is stark. Unions have a major impact on governments, companies, and society more generally. Unions sponsor MPs; green NGOs do not. This area of institutional analysis is grossly under-researched—especially its implications for the design of policy instruments and institutions.

2.7 IMPLEMENTATION, ACCOUNTING, AND ECONOMIC POLICY

In much conventional policy discussion, considerations of biodiversity and the environment are treated as an 'add-on'. Once the conventional micro- and

[22] Greenpeace started life as a Canadian pacifist Quaker Group opposed to nuclear weapons and nuclear weapons testing. The 'green' objective was added later, focused on the very vivid images of seals being culled on the Canadian ice.

macroeconomic problems have been addressed, then the consequences for the environment are considered. An alternative approach is to consider all externalities and public goods on a common basis with all the other goods and services in the economy.

The starting point is to address national income accounts, and in particular GDP. If the objective of economic policy is to maximize GDP, it will exclude important elements of social welfare (Arrow et al., 2004; Helliwell et al., 2012). Consideration has been given to the wider, non-market sources of utility in so-called 'happiness' measures.[23] But what limits the relevance of GDP is that it is a cash-based measure, with no balance sheet. An increase in GDP takes no direct account of assets and liabilities, and changes in their values. There is no explicit allowance for capital maintenance or provision for future liabilities.

This is beginning to change, with substantial efforts by the World Bank (2006) and others.[24] In the UK, the introduction of 'Whole of Government Accounts' (H.M. Treasury, 2011) includes future pension liabilities. What remains is to include infrastructure, both physical and social, and human capital. Environmental assets are part of that infrastructure, with natural capital considered alongside the other forms of capital. The establishment of the Natural Capital Committee is one step towards rectifying this situation.

Valuing natural capital at the net present value of the stream of ecosystem services is in its infancy. The UK National Ecosystem Assessment (UK NEA, 2011), described by Bateman et al. in Chapter 5, is a tentative step in this process, but what remains is to add the natural assets on a case-by-case basis to the balance sheet (see Chapters 8 and 9 by Barbier and Hamilton, respectively). Whilst such an exercise is complex and requires often crude approximations, and will inevitably be built up gradually, it goes in the right direction: it is better to be approximately right, than precisely wrong.

As natural capital is added to the balance sheet, it can be used not just to consider whether and to what extent biodiversity is being preserved, but also to estimate the required capital maintenance as a charge on current spending. We can treat natural assets as 'assets-in-perpetuity' rather than assets that can be depreciated. We want to pass them on to the next generation, to meet a sustainability criterion. The capital maintenance is the sum required to maintain the assets intact, or at least to maintain the value of the service delivered by those assets.

[23] See Layard (2006), and Stiglitz et al. (2009). Furthermore, it is argued that the Aristotelian objective of *eudaimonia*, or 'human flourishing' (Ostwald, 1962), is broader and more fundamental than mere happiness (e.g. Sen, 1993).

[24] See Pearce et al. (1996), Hamilton and Clemens (1999), Arrow et al. (2004), World Bank (2006), Dasgupta (2010), and Arrow et al. (2012), among others. The World Bank has extended its work with the Wealth Accounting and the Valuation of Ecosystem Services (WAVES) partnership.

The final step is to check whether the capital maintenance is being met by current spending on natural assets, either directly by government or by appropriate taxes, subsidies, and permit schemes as described in section 2.6.

2.8 CONCLUSIONS

The maintenance of biodiversity is one of the most complicated resource allocation problems. Biodiversity is heterogeneous, arises at a number of different levels, creating multiple externality and public goods problems, and it almost always poses problems in contexts where there are other multiple market failures too. The tools of economics can help—although they are primitive—in addressing systems properties, over long time periods, within which there is no assumption of stable equilibria. There is no ‘balance of nature’, against which to define optimal equilibria.

So the economic toolbox is not empty. The main tools to hand are: the concepts of public goods and externalities (which tend to disaggregate biodiversity into species and specific habitats); CBA; renewable and non-renewable depletion theory; substitution rules for sustainable development; policy analysis of market-based incentives and regulation; game theory for agreements; and institutional design.

Given the rapid rate of extinction and the collapse of ecosystems on the one hand, and the failures of the main policy instruments and institutions on the other, the scope for policy improvement is enormous. Although the economic tools are imperfect, they are being developed and refined. The practical application of economics to one of the world’s most important resource allocation problems is long overdue.

In order to make progress, the first step is to fully incorporate the natural environment into the economic calculations, and into the core of government accounts. Natural capital needs to be set alongside conventional capital, human capital, and labour, extending the work of Kirk Hamilton[25] and the WAVES partnership, funded by the UK, Japan, and Norway.[26] Such an integrating approach would necessarily overcome the current, all too frequent, assumption of a zero value for natural assets, and requires valuation techniques to be applied.

Once environmental assets are incorporated into national accounts, the next step is to set intergenerational rules. The good news is that the theory has been developing over the last two decades, and the application of intergenerational policy has already begun for the climate, with carbon prices

[25] See Pearce et al. (1996) and and Hamilton and Clemens (1999).
[26] See <http://www.wavespartnership.org>.

gradually emerging and being incorporated into economic policy. Mainstreaming natural capital is required—and with it the mainstreaming of biodiversity. This in turn requires integrating economic analysis into biodiversity policy—and incorporating biodiversity, and natural capital more generally, into the core of economics.

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Part I

Concepts and Measurement

3

Biodiversity: Its Meanings, Roles, and Status

Georgina M. Mace

3.1 INTRODUCTION

A concern for nature and the conservation of the natural world trace back over centuries, but the term 'biodiversity' and some of the concepts it encapsulates are relatively new, tracing back to discussions in the US in the mid-1980s (Wilson, 1988). The word is simply a compression of the two-word term 'biological diversity', meaning essentially the variety of life (Reaka-Kudla et al., 1996). Numerous recent analyses have documented the state of, and trends in, biodiversity, and all conclude that while our knowledge is far from complete, global biodiversity is spectacular, extensive, and widely appreciated. To very many people the rich diversity of life on Earth is the defining feature of our planet. Yet, at the same time, biodiversity is in decline everywhere, largely as a result of a growing human population and the demands for land and resources that result. Concern about the loss of biodiversity also has a long history, but in its recent form traces back to the Convention on Biological Diversity signed in Rio de Janeiro in 1992, from which many concepts in turn have their origins in the Brundtland Report (1987). Pearce and Moran (1994) examined the economics of biodiversity just after the Rio conference. They spelled out clearly how failures to capture the economic values of biodiversity result in economic incentives being stacked against biodiversity conservation and in favour of activities that deplete biological resources. Since then, the same patterns have been observed repeatedly and, if anything, matters have deteriorated further. Biodiversity is not included in economic accounts, because it is a public good, its values are hard to estimate, and impacts of loss are often dispersed or remote from the causal processes. In this chapter I review the growing understanding of what biodiversity is and what it does for people and the rest of life on Earth. I use this to draw some conclusions about how biodiversity might best be reflected in economic analysis.

3.2 WHAT IS BIODIVERSITY?

Beyond its general meaning reflecting the variety of life on Earth, the term 'biodiversity' is now common in a wide range of situations, from ecology, through conservation biology, nature conservation, environmental sciences, and environmental policy. It is used to mean many different things, usually centred on the variety of species in a location (DeLong, 1996), but it can mean all of life on Earth or sometimes, more symbolically, it is perceived to represent wilderness, wild nature, or even natural heritage more broadly, sometimes even including human history and artefacts (Fischer and Young, 2007). In this chapter I will outline common approaches to measuring biodiversity, then describe its roles, state, and trends—and in a way that is relevant for economics. There are many other comprehensive discussions of the definitions of biodiversity dealing with the range of theories and concepts involved (Gaston, 1996; Maclaurin and Sterelny, 2008; Faith, 2013), or with approaches to its measurement (Magurran, 2003).

Biodiversity describes variation among units of life, but the units of biodiversity are themselves many and varied. They include species, genes, populations, communities, biomes, and ecosystems. In this list, genetic diversity is the most fundamental unit, but species richness is used most often. Species are on the whole objective units on which evolutionary pressures act, and they share a common genetic history packaged up into functioning organisms that have evolved, adapted, and interbred in a shared environment. Species lists are relatively straightforward to compile, and resonate with public and specialist interest in natural history. In many ways, therefore, species are the natural units with which to measure biodiversity. Some problems arise because species concepts are variable and fluid (Hey, 2000), they may not work well for microorganisms (Fraser et al., 2009), and can lead to lists containing different numbers of species, with different distributions, depending on the species concepts used (Agapow et al., 2004; Mace and Purvis, 2008; Maclaurin and Sterelny, 2008); but in practice species are practical, biologically meaningful, and widely understood. On its own, however, the species level is inadequate for biodiversity assessment because other biological dimensions vary systematically in ways that are important for biodiversity form and function.

Genetic variation that exists within and between species and populations represents the raw material for structure, form, and function. It is changes in the genotypes (the genetic make-up of an organism) resulting from natural selection acting on genetic variation that lead to the variation in phenotypes (the physical characteristics of an organism) observed in the natural world. Many will argue that the fundamental unit for biological diversity is therefore genetic variation which further enhances the adaptive capacity of living systems (Mace and Purvis, 2008). After discounting genetic diversity shared among species via a common evolutionary history, the entire suite of unique diversity reflected in a phylogenetic tree is the best representation of the overall diversity of those elements (species or populations) represented at the end of the branches of the tree (Vane-Wright et al., 1991). The metric, phylogenetic diversity (Faith, 1992) is a surrogate for disparity or character diversity, and for information content more generally. Character diversity seems likely to be more important for ecosystem function than simple species richness, so maximizing the character diversity conserved has obvious value and can be used for efficient conservation planning.

Populations and communities are significant units below the species level. This is where ecological and evolutionary processes mostly act. Environmental and species interactions within populations and communities comprise a rich and complex suite of dynamics which have a large influence on future abundance and distribution of populations, and hence of species. These interactions, both biotic (involving other organisms) and abiotic (involving the physical environment), drive both the functioning of ecosystems and the fate of species. Some have argued, therefore, that population declines, biomass, and community change are more responsive measures of biodiversity change than species-level metrics, have a greater relevance to ecosystem functions and services, and should take precedence over species extinction rates for monitoring biodiversity change (Hughes et al., 1997; Balmford et al., 2003).

In practice, any effort at biodiversity measurement is faced with enormous problems due to gaps and biases in the information available. Probably less than 10 per cent of all the species on Earth have been described and named, and what is known is strongly biased towards vertebrates, terrestrial, and temperate areas. Some of the most numerous and diverse taxa, such as the invertebrates and fungi, are extremely poorly studied, and estimates of the total number of species are still very uncertain (Costello et al., 2013).

Given the difficulty of identifying, counting, and classifying species, studies are increasingly replacing taxonomic classifications with analyses based on units that reflect structural and functional groupings. For example, estimating the abundance of trees versus crops is relatively straightforward compared with counting all the component species in an area of forest versus farmland, and can provide a practical means to measure structural diversity in a landscape and its change over time. Functional groups of organisms also allow extrapolations to ecosystem functioning—for example, examining trends in the distribution of decomposers versus consumers, or plants with relatively large versus small leaf areas, might represent high-turnover or high-productivity areas, respectively. Other functional groupings might represent the habits of different species and potentially their vulnerability to, or impact upon, people. For example, without knowing all the species individually, a biological community can be examined to measure the biomass of predators compared to herbivores, or abundance of species that are good invaders compared to species that are strong competitors. These kinds of classifications of biodiversity based on structural and functional traits are gaining popularity because they are comparatively tractable and allow extrapolations even with limited data. Moreover, certain trait classifications allow for models and maps to be developed that are useful for assessing biological community functions (Lavorel and Garnier, 2002), modelling responses to anthropogenic pressures (Purves et al., 2013) and with Earth system models (Kattge et al., 2011), especially for the interactions between the biosphere and the climate system (De Deyn et al., 2008). They are also the norm for assessing the diversity of microorganisms where the usual concepts for species, populations, and even individuals break down. Increasingly, as the functional roles of species and ecosystems take on greater significance in arguments for conservation, traits and functions may start to eclipse the need for comprehensive identification of species, although on their own such measures may miss important diversity elements. For example, the definition of traits is often subjective or idiosyncratic, and trait diversity does not then represent phylogenetic diversity.

Finally, to avoid the difficulties of enumerating species or groups of species, some recent assessments simply consider the status of geographically defined areas such as biomes, habitats, or ecosystems. These are all different approaches to classifying distinctive areas of land or sea, distinguished by the dominant biota as well as the underlying physical environment and biogeographical history. WWF has defined over 800 'ecoregions' worldwide. It defines an ecoregion as 'a large unit of land or water containing a geographically distinct assemblage of species, natural communities, and environmental conditions'. The ecoregions are mapped and species lists are compiled for them (Olson et al., 2001), so they provide a practical unit for global analysis of the extent of pressures and environmental change affecting areas with different amounts of species-level biodiversity. Each ecoregion is unique, but they are further classified into twenty-six major habitat types, sometimes called biomes. These describe different areas of the world that share similar environmental conditions, habitat structure, and patterns of biological complexity, and contain similar communities and species adaptations. For example, two biomes are the Tropical and sub-tropical moist broadleaf forests, and Deserts and xeric shrublands. Biomes are practical units for assessing broad patterns of biodiversity change globally (Lindenmayer et al., 2012). Further, in order to represent the unique fauna and flora of the world's continents and ocean basins, each major habitat type is further subdivided into seven biogeographic realms (Afrotropical, Australasia, Indo-Malayan, Nearctic, Neotropical, Oceania, Palearctic). Analyses can then be undertaken across major biogeographical zones, across major habitat types, or both, and this approach has been effective for assessing status and trends in poorly studied groups of plants and animals that would not otherwise be represented, especially non-vertebrates.

Ecoregion- and biome-based analyses provide information on the composition and diversity in different areas, but alone these are not enough to inform about biodiversity processes. Processes are both a cause and consequence of biodiversity in a particular location. Ecosystems are structured in many ways, reflecting history, process, and function. On its own, biodiversity is an outcome of physical and biological processes that have tended over time, and in the absence of major perturbations, to increase diversity. Ecological and evolutionary processes, playing out on a biogeographical stage, generate the variety and composition to be found in any one place. In recent times the major agent of large-scale perturbations has been the growing size, distribution, and impact of people on the Earth. Recent impacts (over decades to centuries) have resulted in rates of biodiversity loss orders of magnitude higher than average rates in pre-human times, that approach rates seen in the most dramatic mass extinctions of the palaeontological past (Barnosky et al., 2011). However, different components of biodiversity are being lost at different rates; changing composition and loss of extent and biomass in major biomes are now much more marked than simple loss of diversity (Pereira et al., 2012). Modelling approaches that link patterns in the turnover of biological richness to spatial landscape units as a means to assess biological change more generally are now being developed and used, building on the growing availability of species records and tools for spatial mapping of the landscape (Ferrier et al. 2004). Such approaches provide useful trend information for both changes in biodiversity pattern and process, though the link to recognizable biodiversity units is lost.

Different disciplines favour different measures of biodiversity. Ecologists tend to think about biodiversity in terms of the forms and functions of organisms in a place, especially in a community or an ecosystem, because it is the structuring of varieties in space and time that leads to functions and dynamics that they seek to understand. Evolutionary biologists similarly think about the dynamics, but with an increasing focus on the historical or inherited variation, and therefore the genetic and phylogenetic attributes. Conservation biologists are sometimes concerned with function and process, as they should be, but often also with preservation of species or genetic diversity, seeking efficient and achievable solutions to the allocation of limited resources. For nature conservationists and wildlife managers, biodiversity often simply means the maintenance of wild habitats and species.

3.3 MEASURING BIODIVERSITY

The discussion to date shows that there are very many dimensions of biodiversity (e.g. composition, function, structure). How then can all this complexity be measured, and indeed should we aim to measure it all, comprehensively or integrally? Proposals have been made to measure composition, structure, and function, independently in a nested hierarchy that incorporates each one at four levels of organization: regional landscape, community-ecosystem, population-species, and genetic (Noss, 1990). Clearly, the definition and measurement of biodiversity can then become very complicated, and can lead to requirements that greatly exceed the limited knowledge base. Even having done this, it is not clear what the result could be used for; how the different dimensions and levels should be weighted, and if it is really useful if some iconic or crucial element is lost entirely but the overall statistic shows little change. The problem of measuring biodiversity is not one that can be addressed by comprehensive suites of metrics, which quickly become too complicated, or by a single, composite metric, that attempts to capture all possible measures of interest. Despite many attempts to develop a composite measure of biodiversity, the task is doomed to failure. In almost all cases it simply confounds different metrics that represent different attributes, and the interesting and important detail is easily lost.

Because it is so impractical to think we could ever enumerate all of these measures, simple metrics, such as the number of species in a place, are most often used as indicators of biodiversity, despite their evident inadequacies. Many legal and policy instruments rely on species lists and other measurable aspects, even though these are themselves incomplete and unrepresentative. Thus, for example, the primary datasets reported by national governments tend to rely heavily on bird, butterfly, and flowering plant species recording that is largely supplied by naturalists and NGOs. Any attempt at comprehensive species monitoring faces the problem of data gaps and biases, though new coordinated databases such as the Global Biodiversity Information System (GBIF), spatial modelling approaches (Ferrier et al., 2004), the emergence of new networks such as GEOBON (Scholes et al., 2008), online efforts to integrate datasets (Jetz et al., 2012), and new sampling approaches (Baillie et al., 2008), mean that progress is now being made with available data.

It is clear that we need to design biodiversity observation and measurement systems better (Scholes et al., 2012), but this still begs the question of what the measures should be better for. Of course, the most effective approach is to define the questions about changes in biodiversity first, and then design the monitoring, measurement, and research that specifically addresses the questions at hand (Green et al., 2005; Mace and Baillie, 2007), but even this apparently focused approach may often lead to a large suite of metrics.

Pereira et al. (2013), for example, suggest five classes of essential biodiversity variables needed for global monitoring of biodiversity change (genetic composition, species population abundance and distribution, species traits, community composition, ecosystem structure and ecosystem function). Each of these may have several different metrics, reflecting different places or groups of organisms, over temporal scales ranging from one year to several decades.

This section has illustrated the complex nature of biodiversity, the many different perceptions of what it involves, and the problems that arise in determining how to measure it, especially to assess change. Solutions start to flow more quickly when addressing a narrower set of issues, or better still, asking specific questions that can then focus the measurement more narrowly. In the next session I focus on biodiversity as defined by the UN Convention on Biological Diversity (CBD), and use the CBD's goals and targets in 2010 as a basis for defining biodiversity, measuring its trends, and using this information to assess the consequences of biodiversity decline for people and their welfare.

3.4 A WIDELY USED DEFINITION OF BIODIVERSITY

The Convention on Biological Diversity, established in 1992, adopted a broad, inclusive, but biologically based definition that has proven useful for many purposes:

> 'Biological diversity' means 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, Article 2)

The CBD definition has several features; it makes the point that diversity can be anywhere in land, sea, or freshwater, that the diversity can be within species (so including genetic diversity), between species, and above the species level, including ecological communities. It also includes the diversity of ecosystems. This is a slightly curious level at which to observe biodiversity because ecosystems come in a very wide variety of scales and types, ranging from a single small pond to an entire ocean, or a patch of soil to an entire savannah or prairie. Ecosystems are also recognized to include both abiotic and biotic components. Biodiversity is a part of an ecosystem, and by this definition, ecosystems are part of biodiversity. In most usages where the level of organization above the species has been used it appears as habitats or biomes, usually as defined in the WWF classification (see earlier), and the variety of these can be catalogued and monitored over time. There are two other features of the CBD definition that cause confusion. One is that it includes reference to the 'ecological complexes' of which species are part, presumably reflecting the interactions among species and community-level processes. From an ecological and evolutionary perspective this is important; ecosystem functions and processes are mostly a consequence of interaction and dynamics, not simply of the standing stock of organisms and species. Second, the CBD definition is *only* about variability. This makes it a diversity-only definition. However, in many common usages the loss of biodiversity means the loss of area, biomass, or amount, rather than the loss of variation. Thus, to report that 10 per cent of forest area was lost could not be used to mean that 10 per cent of forest diversity was lost. Mostly this means just a loss of area, and although diversity increases with area, the relationship is allometric—even a large proportional loss of habitat, such as 50 per cent, may leave more than 90 per cent of species remaining, and for small proportional losses of habitat there will be much smaller losses of species. To look at it another way, 50 per cent of global bird species richness can be captured in just 2.5 per cent of global land area (Orme et al., 2005), and the same pattern is evident in many other species groups.

Biodiversity is not the right term to use to reflect the changing state of nature overall, which is better reported using metrics related to population size, numbers of populations and habitat extent, instead of diversity (Balmford et al., 2003). Despite these small difficulties, the CBD definition is widely used and is sufficiently inclusive to cover most needs. Most significantly, it has led to a series of policy goals and mechanisms developed by the Parties to the CBD (Table 3.1). The CBD strategic plan for 2011 to 2020 presents a coordinated set of goals and targets, which aim to embed biodiversity conservation in wider societal value systems, reduce direct pressures on biodiversity, safeguard species and ecosystems, ensure benefits from biodiversity, and support the provision of resources. There are some significant challenges in achieving these targets that will require concerted efforts from both biodiversity scientists and natural resource economists working together. For example, targets 2, 3, and 4 require reform and redesign of policies and subsidies in order to ensure sustainable flows of resources while maintaining the system within safe limits. Targets 7 and 11 call for full accounting of biodiversity considerations in production sectors, and the secure management of genetic resources.

The Aichi targets in Table 3.1 present a clear agenda for global efforts to maintain biodiversity. But there are potential conflicts among targets that will become more apparent once the good, broad intentions are translated into practical action at and below country level. At this point it will be necessary to consider in more detail what the roles of biodiversity are, when and where it matters, and what aspects are more or less easy to forgo.

Table 3.1. Goals and targets agreed by the 10th meeting of the Parties to the Convention on Biological Diversity

Strategic goal A: Address the underlying causes of biodiversity loss	Target 1: By 2020, people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably
	Target 2: By 2020, biodiversity values are integrated into national and local development and poverty-reduction strategies and planning processes and national accounts
	Target 3: By 2020, incentives, including subsidies, harmful to biodiversity are eliminated, phased out or reformed
	Target 4: By 2020, governments, business and stakeholders have plans for sustainable production and consumption and keep the impacts of resource use within safe ecological limits
Strategic goal B: Reduce the direct pressures on biodiversity and promote sustainable use	Target 5: By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced
	Target 6: By 2020 all stocks managed and harvested sustainably, so that overfishing is avoided
	Target 7: By 2020 areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity
	Target 8: By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity
	Target 9: By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment
	Target 10: By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning

(*continued*)

Table 3.1. continued

Strategic goal C: To improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity	Target 11: By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas are conserved through systems of protected areas
	Target 12: By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained
	Target 13: By 2020, the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives is maintained
Strategic goal D: Enhance the benefits to all from biodiversity and ecosystem services	Target 14: By 2020, ecosystems that provide essential services, including services, are restored and safeguarded
	Target 15: By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems
	Target 16: By 2015, the Nagoya Protocol on Access and Benefits Sharing is in force and operational
Strategic goal E: Enhance implementation through participatory planning, knowledge management, and capacity building	Target 17: By 2015 each Party has developed, adopted as a policy instrument, and has commenced implementing an effective, participatory and updated National Biodiversity Strategy and Action Plan (NBSAP)
	Target 18: By 2020, the traditional knowledge, innovations, and practices of indigenous and local communities and their customary use, are respected
	Target 19: By 2020, knowledge, the science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved, widely shared and transferred, and applied
	Target 20: By 2020, the mobilization of financial resources for effectively implementing the Strategic Plan for Biodiversity 2011–2020 from all sources should increase substantially

Note: These so-called Aichi targets were agreed by over 180 nations present in Nagoya, Japan, in October 2010, to represent the Strategic Plan for Biodiversity 2011–2020.

Source: Convention on Biological Diversity, Aichi Biodiversity Targets.

3.5 WHY DOES BIODIVERSITY MATTER?

There are a wide range of answers to this question, which I will summarize here only very briefly as they are addressed elsewhere more fully (Faith, 2013).

3.5.1 Extrinsic and intrinsic values

It is useful to consider the values that biodiversity holds for people across a continuum, from use to non-use values (using the Total Economic Value typology). In addition, however, it is important to acknowledge that for some people their values lie outside of this spectrum of extrinsic values, and are intrinsic. Intrinsic value refers to the view held by many people that the natural world, and therefore biodiversity, merits conservation for reasons beyond any material benefits or measurable values. According to this view, the intrinsic value of nature cannot be compared with any other value set, and therefore proponents of this view not only find valuation unacceptable in principle, but also cannot countenance comparisons or priority-setting among different components of nature. Although some people consider that intrinsic values are captured through stated preferences and option values, this is contested.

Intrinsic value should not be confused with the various kinds of extrinsic, non-use, non-market values such as option, existence, and bequest values. These are difficult to estimate but dominate many people's concerns for the conservation and protection of biodiversity and ecosystems, and there are various techniques available for obtaining relative measures of value, even if these are not very robust and impossible to tension against monetary values (see Chapter 6 by Atkinson et al.). Use values include biodiversity contributions to both direct and indirect values. Direct values are provided by biodiversity that contributes to products and processes such as for food, pharmaceuticals, and chemicals. Here there is a market, and market values can be established. Non-use values are provided by various functions and services underpinned by biodiversity, which include many public goods and assets for which markets do not exist. Examples include pollination, pest regulation, clean water, and recreational values. The use/non-use typology has been very influential, but for biodiversity it has been largely taken over in recent years by the emergence of the concepts of ecosystem service, and its increasing application in both science and policy (Millennium Ecosystem Assessment, 2005*a*).

3.5.2 Ecosystem services

Ecosystem services are the benefits people obtain from ecosystems. As defined by the Millennium Ecosystem Assessment (2005*a*), these include provisioning services such as food, water, timber, and fibre; regulating services that control climate, floods, disease, wastes, and water quality; cultural services that provide recreational, aesthetic, and spiritual benefits; and supporting services such as soil formation, photosynthesis, and nutrient cycling. This classification has been revised recently to facilitate economic valuation. The main changes have been to remove supporting services as a category, since they are really fundamental ecosystem processes that underpin most other services, and to separate ecosystem functions and processes from 'final ecosystem services' which provide goods to people. The final ecosystem services are characterized in the ecosystem, but the goods which have measurable values are in the wider economy (Fisher et al., 2008; Bateman et al., 2011).

Biodiversity and ecosystem services are often bracketed together as if they are the same thing, and there is an interesting history about how the two concepts have co-evolved over the past twenty years (Lele et al., 2013). In the Millennium Ecosystem Assessment (2005*b*), biodiversity is presented at the core, as one foundation of all ecosystem services, but it is also described in many places as an ecosystem service itself, and as being an 'enabler' or regulator of ecosystem services (Díaz et al., 2006). At the same time, existence value and many of the non-use values of biodiversity are represented as being one significant type of cultural ecosystem service. The literature is further confused by frequent use of the phrase 'biodiversity and ecosystem services', linking the two together as if they are in some way distinct yet completely linked. None of these relationships is tenable on its own. Most ecosystem services rely on physical and chemical inputs as well as biological inputs, and many biological inputs to ecosystem services do not depend primarily on diversity. Some ecosystem services are enhanced with a reduction in biodiversity (Cardinale et al., 2012)—for example, food production, which is one of the successes of agricultural intensification. The conservation of diversity will therefore not necessarily maximize overall ecosystem service delivery, especially over short time scales.

Mace et al. (2012) present a typology for the different ways that biodiversity and ecosystem services are related. Here biodiversity can be (1) a regulator of underpinning ecosystem processes; (2) a final ecosystem service; or (3) a good that is subject to valuation. The first of these equates to the role of biodiversity in ecosystem processes and functions, which is itself an area of continuing active research and debate in ecology (Cardinale et al., 2012). The underpinning roles of biodiversity ecosystem functions and processes include biodiversity contributions to primary production, decomposition, nutrient cycling, as well as pollination and disease resistance. These roles are performed primarily by microorganisms in soil and water, and by invertebrates and plants. The second role concerns the extent to which diversity is itself of value to final ecosystem services—for example, in bioprospecting, or for crop and livestock varieties. The third relates to cultural values that come from wild nature and ecosystems—primarily the enjoyment, inspiration, and aesthetic pleasures that people derive from nature, including striking diversity in, for example, coral reefs or the tropical rainforests, or from seeing rare and charismatic species. Interestingly, the types of species and the biodiversity metrics vary widely among these three levels. While the contribution to ecosystem processes is largely from plants and microorganisms, and trait diversity seems to be a key metric, the cultural values are largely from large-bodied, charismatic birds and mammal species. Here rarity and distinctiveness are important. At the level of goods, effectively the direct-use values, the fundamental metric is probably genetic diversity, essentially a source of evolutionary novelty. This three-way distinction is one way to view the complicated relationship between biodiversity and the benefits people derive from it. Understanding this has important implications for both conservation and ecosystem management (Mace et al., 2012) where different biodiversity components will have different values.

As already suggested, and as is obvious from brief reflection on landscapes that have been modified for production or for some regulating services, the diversity of genes, species, and traits is not always correlated with high ecosystem service delivery. Food production has been enhanced by breeding selectively for particular strains with low diversity that can reliably produce high yields. Grasslands managed for flood control have low diversity of species; coastal dunes rely on a few species that are able to grow extensive root systems in sand in order to protect the coastal strip from erosion. However, the same is not true of biodiversity and ecosystem function relationships. Cardinale et al. (2012) undertook a systematic review of research that has examined how biodiversity loss influences ecosystem functions, and showed that almost without exception, biodiversity in terms of genes, species, or trait diversity positively increases the efficiency with which ecosystems capture and convert energy, and decompose and recycle organic material. These are the most fundamental aspects of ecosystems, and ones on which people ultimately depend for energy and nutrients. In addition, again in most cases studied for biodiversity–ecosystem function relationships, more diversity enhances stability and resilience. In contrast, a simple meta-analysis across multiple studies showed that the relationships between biodiversity and ecosystem services are often not positive, are sometimes mixed, and are often hard to predict. This is partly because some ecosystem services depend less on biological components in the environment, and more on physical and chemical components, and partly because for some services, efficiency is improved with low diversity. These studies emphasize the important balance to be achieved between long-term resilience supported in diverse ecosystems, compared with short-term high production achieved in low-diversity systems. This is a critical area where agricultural and biodiversity scientists need to work more closely together.

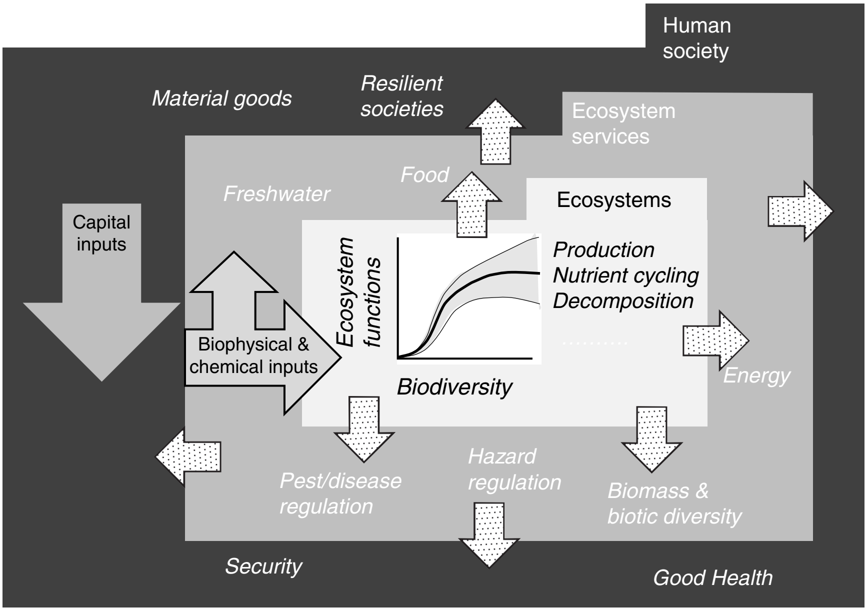

Fig. 3.1. Schematic representation of the role of biodiversity in supporting ecosystem functions and wider societal benefits from ecosystems

Figure 3.1 is a schematic representation of the way that biodiversity underpins ecosystem functions and services. It illustrates the tighter relationships between biodiversity and ecosystem functions which then underpin other services and benefits, but with increasing moderation by other factors.

Within ecosystems, fundamental processes require biotic inputs, and a positive relationship affecting biodiversity-to-ecosystem functioning is common. Ecosystem functions, such as production, nutrient cycling, and decomposition then support ecosystem services. Ecosystem services generally require other forms of biophysical inputs, as well as capital inputs for infrastructure and production systems. The biodiversity–ecosystem service relationship is therefore a little weaker than the biodiversity–ecosystem function relationship. However, resilience and security are also increased with higher levels of biodiversity. Ultimately, the core needs of human society, such as material goods, energy, security, and resilience, are underpinned in a range of ways by biodiversity, but the significance of other forms of capital inputs increases from the core areas of the diagram outwards.

There are two other key reasons to care about maintaining biodiversity, both of which contribute to ecosystem services but which also merit consideration in their own right. These are the contribution of biodiversity to heritage, adaptability, and resilience, and the significance of biodiversity in representing the complete genetic library of life. I turn now to a discussion of each of these.

3.5.3 Heritage, adaptability, and resilience

Current pressures from a rapidly growing human population and the intensifying demands for consumption are placing a huge strain on the world's landscapes and seascapes. At the same time, environmental change, including climate change, is resulting in species and ecosystems facing rates and intensities of change greater than at any time in their recent history. These natural systems have a range of adaptive mechanisms at their disposal. Evolution, dispersal, and adaptive radiation have allowed natural communities to develop, fill new niches, and adapt to challenges in the past. But people now dominate the Earth, natural habitats are reduced and fragmented, and dispersal may be a much more limited option than it was in the past due to the loss and fragmentation of most natural habitats. The raw material for adaptation is genetic diversity, structured in populations, distributed across species ranges in interactions with other species and different niche conditions. Without doubt, less diverse populations and communities will fare worse in the future than more diverse ones. Loss of genetic diversity at the level of individual organisms, within populations, or across species ranges, all compromise the potential for adaptation. Loss of entire species represents the loss of millions of years of adaptive evolution that can never be replaced. The genetic variability represented in life on Earth is therefore an immense genetic library that forms a source of resilience, and is our heritage and our responsibility.

It is at this point that the utilitarian needs for biodiversity come face to face with biodiversity conservation.

3.6 CONSERVATION OF BIODIVERSITY

Concern for nature has a long history, but as currently practised became common in the twentieth and twenty-first centuries. The term conservation, as opposed to preservation, is quite recent, becoming established only in the past fifty years or so. Preservation was characteristic of colonial regimes and implied stasis, whereas conservation implies rational use (Adams, 2009). The differences between rational use and more preservationist concerns have remained in tension ever since—for example, in the debate between those who argue that conservation is most effectively based on the sustainable use of resources and those who argue for preservation, and between those who argue on behalf of conservation versus those who favour rural poverty alleviation.

The driving concern for conservation is usually expressed in terms of loss of species, but as discussed earlier, defining metrics is far from simple, and information is sparse and disorganized. Most commonly, organizations and governments use measures based on information that is available, and this is often a poor sample of what exists. Until very recently, available information for conservation assessments was dominated by species lists, most often concentrating on the vertebrates, especially birds and mammals, but sometimes including butterflies, trees, or well-studied groups of flowering plants. This is very far from a comprehensive sample of all biodiversity. These groups themselves comprise much less than 10 per cent of described species, and little more than 1 per cent of the total. More recently the availability of remotely sensed information and the compilation of shared species data has led to burgeoning information on biomes, land use, and major habitat types, that generally complement species lists (Butchart et al., 2010).These data are useful at large scales for overviews and syntheses of status and trends, but more local information appropriate to conservation planning on the ground remains patchy and incomplete, with an unhelpful bias towards better information in the least diverse areas (Collen et al., 2008).

Recent assessments have also emphasized the distinction to be drawn between biodiversity loss (generally species extinction, but some loss of genetic variation as local populations lose range extent and abundance) and biodiversity alteration (changes in abundance and community structure, range shifts) (Pereira et al., 2012). Conservationists are concerned about biodiversity alteration because a range shift can be a local extinction, and community-level changes can have consequences for ecosystem stability and function. Biodiversity alteration is reversible (at least to a degree), while biodiversity loss (with current conservation interventions at least) is not; in principle habitats can be restored and local species populations recovered, while species extinction is for ever. To date, biodiversity alteration has been far more significant than biodiversity loss, especially at the species level, but future projections lead to the conclusion that rates of loss must increase (Pereira et al., 2010).

Conservationists often talk about the importance of conserving not only species and ecosystems, but also the evolutionary processes that formed them. This objective to retain the potential for species to respond to natural selection through evolution is likely to become more significant in future as environments and their pressures change at ever increasing rates and intensities. Conservation is therefore not simply aiming to retain all current species as if they were books in a library, but is seeking to maintain the elements from genetics, environment, and natural selection that will allow future species to persist and diversify, or analogously for new books to be written. Thus, conservation planning requires networks of interacting populations preserved in a coherent set of sites where habitat protection and species conservation are a primary goal of management.

Conservation plans directed at species or at habitats, and habitat conservation are most effectively pursued through protected areas. The World Commission on Protected Areas (WCPA) defines ‘protected area’ as:

> a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values. (Dudley, 2008, pp. 8–9)

Protected areas need to be well managed to be effectively conserved, and protected area systems need to be distributed across the full range of ecosystems—terrestrial, freshwater, and marine—to be fully representative. However, protected areas can become isolated and, once surrounded by other forms of land use (a forest surrounded by agriculture, for example), they will lose species. Greater isolation leads to increasing rates of degradation (Boakes et al., 2010), and reserves are increasingly becoming isolated in a matrix of intensively managed land (DeFries et al., 2005). These kinds of concern have been matched by developments in the field of 'landscape ecology' and a growing literature on the possibility of creating connections between ecosystem fragments, along which species might move easily, developing further the ecology of linked or 'meta' populations. There is increasing interest in the idea that conservation should be pursued through sets of protected areas managed as part of ecological networks in landscape-scale conservation.

Though there is no single agreement on what it means to conserve a species—other than to keep it from becoming extinct—recent work has proposed six attributes of a successfully conserved species. The species should be: (1) demographically and ecologically self-sustaining; (2) genetically robust; (3) have healthy populations; (4) have populations distributed over the full ecological gradient of the historical range; (5) have more than one population in each of these ecological settings; and (6) be resilient to environmental change (Redford et al., 2011). This list might be regarded as the successful endpoint of species conservation, and is clearly far more than simply ensuring the survival of those species outside threatened species lists, such as those maintained by the IUCN and recorded in Red Lists.

A recent trend in conservation is to move from policies that are geared to the avoidance of undesirable outcomes (e.g. species extinction) towards plans with positive goals reflected in systematic conservation planning (Margules and Pressey, 2000), and to integrate these into wider goals for management of natural resources on land and in the sea. Thus, the CBD targets for 2020 (Table 3.1), for example, include species and habitat conservation (Targets 11 and 12) within a broader framework for the overall maintenance of biodiversity for the benefit of people and all of life on Earth.

3.7 CONCLUSIONS

Biodiversity is not a simple concept. It embraces not only a wide range of biological attributes and functions, but it also means different things to different people, and its value is almost always going to be context-dependent. If economics is the science that analyses the production, distribution, and consumption of goods and services upon which wealth and welfare depends, then it will be necessary to disaggregate the components of biodiversity that influence and are influenced by people's wealth and welfare.

If biodiversity is going to be successfully conserved for the benefit of people and of all life on Earth, then its value must be fully incorporated into decision-making. Having no value, or holding arbitrary values, cannot support decision-making that will break the loop whereby economic forces continue to drive the extinction and loss of biodiversity, even though it is clear that it has value and is valued. The starting point for economic valuation must come from accounting properly for the benefits that flow from biodiversity. I listed these earlier in the general categories of intrinsic and extrinsic values, ecosystem services, heritage, adaptability, and resilience. These are overlapping categories, but different kinds of classifications are appropriate to different contexts. For example, for land-use planning and for achieving the successful integration of biodiversity conservation into the production sectors, then an approach based around the valuation of ecosystem services is useful (Bateman et al., 2011) as long as longer-term considerations are not neglected. This is appropriate for near-term decision-making, but longer-term considerations for adaptability and resilience depend more on adequate stocks of different biodiversity components, including genetic, community, and ecosystem features. Natural capital accounting and inclusive wealth measures may then be relevant (Dasgupta, 2010) though there remain many uncertainties about thresholds and limits in these systems (Scheffer and Carpenter, 2003; Barnosky et al., 2012). Conservation, recreation, and cultural values tend to be dominated by a subset of species and habitats; rational and efficient conservation planning at national and international level should be able to incorporate both pattern and process if well designed (Pressey et al., 2007).

Identifying the important endpoints from biodiversity for well-being, resilience and adaptability will simplify and focus the identification of the relevant metrics, provide means for more accurate valuation, and should, in time, support the conservation of biodiversity in all its important forms and functions.

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4

Identifying and Mapping Biodiversity: Where Can We Damage?

Kathy J. Willis, Marc Macias-Fauria, Alexandros Gasparatos, and Peter Long

4.1 INTRODUCTION

Where can we damage? Given the increasing global pressures on land for food, fuel, industrialization, and population growth, this question is pertinent for large areas of terrestrial (and marine) landscapes. Globally, unprotected terrestrial landscapes currently account for 87 per cent of the Earth's surface and >99 per cent of the marine realm. It is in these unprotected areas that the vast majority of concessions are granted for conversion of land from 'natural' to 'other' uses on a daily basis. In Europe alone it is estimated that around 1,500 ha of 'natural' land are lost every day to changes in infrastructure and urbanization.[1] Yet it is also widely acknowledged that not all unprotected land is of equal value in terms of the biodiversity it contains and the ecosystem services that it provides—hence the question: 'where can we damage?'

The answer to this question depends on who is asking it. For the biodiversity conservationist, the answer is probably nowhere; all biological diversity has some interlinked function to an ecosystem that is almost certainly important to human well-being. But the luxury of conserving all biodiversity is not an option, especially beyond protected areas, and is often a long way down the list of considerations, if it is on it at all. However, a number of high-profile environmental disasters, (e.g. the BP Deepwater Horizon Oil Spill, Gulf of Mexico, 2011) have recently highlighted the reputational and associated economic risk that can occur if biodiversity is damaged. The question of 'where can we damage?' therefore needs to be answered in an objective and quantitative way, but with a pragmatic understanding that some parts of the

[1] See <http://ec.europa.eu/environment/nature/info/pubs/docs/nat2000newsl/nat27_en.pdf>.

landscape will be changed and these need to be the areas that contain the least important biodiversity.

But how close are we to being able to assess the relative value of biodiversity across landscapes outside of nature reserves? There are a plethora of methods and metrics currently used to measure biodiversity and associated ecosystem services (see Chapter 2). However, even the use of the word 'biodiversity' can represent a whole range of metrics from genetic diversity through to populations and communities. But what biological data do we need in order to identify areas of high (and conversely low) biodiversity across landscapes outside of protected areas? What datasets and models are available to do this? And where are the knowledge gaps? This chapter aims to address these three questions. Section 4.2 considers what data is most appropriate for determining important areas for biodiversity outside of protected areas, and at what spatial and temporal resolution. Different types of biological data currently available and the models and algorithms which can be used to gain an understanding of their spatial configuration across landscapes are then discussed in section 4.3. A brief review is then made of the current suite of tools available to policy-makers and stakeholders to help determine spatial patterns of biodiversity in section 4.4. Finally in section 4.5 we discuss the knowledge gaps and possible future approaches to mapping and displaying the complexity of biodiversity and its links to human well-being across ever-changing landscapes.

4.2 WHAT INFORMATION ON BIODIVERSITY IS NEEDED TO DETERMINE IMPORTANT REGIONS OUTSIDE OF PROTECTED AREAS?

What biodiversity needs to be determined outside of protected areas—richness, abundance, endemism, or rarity? And should it be measured at the genetic level, species, populations, or communities? These questions come back to the very crux of the definition of biodiversity, and arguably a different interpretation needs to be accepted for regions beyond protected areas. As discussed in Chapter 2, the use of the word 'biodiversity' can represent a whole range of metrics, from genetic diversity through to populations and communities. Such broad definitions do not, however, provide much guidance or direction to those who are challenged with trying to determine the different aspects of biodiversity on a landscape and the associated risks if damaged. In the past eight years, the move to focus on the measurement of biodiversity that has direct benefit to human well-being through the ecosystem services that it provides (Millennium Ecosystem Assessment, 2005) has led to the emergence of the term 'biodiversity science'. And it is in the definition of this term that some guidance as to what to value starts to emerge. Biodiversity science is usually defined as an:

> integrative science, linking biological, ecological and social disciplines in an effort to produce socially relevant new knowledge for the conservation and sustainable use of biodiversity.[2]

Using this definition, therefore, it should be biodiversity that is 'socially relevant', with the focus on those aspects of biodiversity that contribute to human well-being through its sustainable use.

So what information is needed to determine 'socially relevant' biodiversity—i.e. the relative ecological value (in terms of its benefits to human well-being) of one area compared with another? This is a question that has been recently addressed by de Groot et al. (2010), specifically focusing on valuation of ecosystem services, although it is arguably equally relevant to all aspects of biodiversity mapping beyond protected areas. In this they state that before any land can be valued for the ecosystem services that it contains, two types of information are required relating to (i) the ecological properties of the landscape; and (ii) the features for supporting ecosystem functions. What this actually means 'on the ground' is discussed in the following section along with a consideration of the most appropriate spatial and temporal scales.

4.2.1 Ecological properties of a landscape

The ecological properties of a landscape are the static features seen presently. These include many of those listed earlier—e.g. current distribution/abundance of species, threatened species, endemics, genetic diversity—with an ideal scenario being one where it is possible to determine the parts of the landscape that have the highest concentration of as many of these features as possible. This is generally the approach used for the majority of schemes to classify important biodiverse landscapes (and seascapes) at global, regional, and local scales in order to determine the placement of reserves and protected areas.[3] Usually such schemes are primarily aimed at determining the most important regions for conservation of biological diversity. However, if the focus is also upon determining areas of biodiversity that are important for their ecosystem services, then the metrics for measuring ecological properties will possibly be different but not completely so (Turner et al., 2007). There are also regions where human well-being is reliant upon one or two key species or vegetation types—for example, economically important plant species. Here priority in mapping ecological features will usually focus on the relative abundance and

[2] See <www.diversitas-international.org>.
[3] See World Database of Protected Planet at <http://www.protectedplanet.net/>.

biomass of a few selected species across the landscape rather than overall richness (e.g. Lobell et al., 2011).

4.2.2 Ecological features of a landscape

In comparison to the 'static' ecological properties, the ecological features of a landscape are those that are important for maintaining the ecological processes and ensuring sustainable biodiversity in space and time. Within an ecosystem services framework, these typically include nutrient cycling, clean water provision, and soil erosion protection. Standing carbon stocks and the role they provide in regulating atmospheric carbon dioxide also come into this category. Within a more general biodiversity framework it can be argued, however, that key ecological features should also include those that enable persistence and sustainability of biodiversity across landscapes. Examples of such ecological features include: (a) those that support connectivity across the landscape (e.g. rivers, wetlands); (b) habitat integrity (i.e. patch size of vegetation); and (c) resilience (i.e. landscape features that maintain a functioning habitat despite environmental perturbations).

Features that support connectivity are particularly essential across fragmented landscapes because they enable the flow of species and associated genetic material (Bengtsson et al., 2003) via migrations and range shifts. Such movements can also be an important dispersal mechanism for other species. For example, species including birds, bats, and large herbivores intentionally or unintentionally transport micro-fauna and flora across landscapes. Large herbivores often carry seeds on fur or in their gut, which then are transported to new locations (Couvreur et al., 2004). The role of these animal transportation systems can have a significant impact on ecosystem functioning in both time and space (Sekercioglu, 2006). Ecological features across landscapes that enable connectivity include river corridors, wetlands, and marshlands (Lundberg and Moberg, 2003). Waterfowl, for example, serve as a long-distance dispersal vector for many aquatic invertebrates. Removal of wetlands can therefore have a much wider ecological impact than just the loss of biodiversity in the immediate water body (Amezaga et al., 2002).

Habitat integrity (i.e. patch size) is another important feature for maintaining overall biodiversity of an area. Numerous studies have demonstrated that the smaller the habitat patch size (i.e. the more fragmented a landscape becomes), the fewer the number of species. This species–area relationship is related to a number of factors, but one of the main explanatory variables is the increased extinction risk associated with smaller population sizes (for review see Triantis et al., 2012). There are also fewer niches available in smaller habitat patches and a greater distance between patches resulting in less connectivity across the landscape. Habitat patches have therefore often been likened to islands—the more remote the island (habitat patch), the fewer species it contains. Within a conservation framework/protected area, it is the fragmented landscape that often therefore takes priority, on the basis that this is a landscape where species are at highest risk of extinction (Harte and Kitzes, 2012). However, for landscapes beyond protected areas, it is arguably the intact patches of habitat that are a more important feature because of the potential that they contain for maintaining a diverse environment and habitat heterogeneity (Willis et al., 2012).

Resilience is the ability of a landscape and the biotic features that it contains to withstand environmental perturbations (Groffman et al., 2006). To give an example, a resilient area is somewhere that when an environmental perturbation hits (e.g. hurricane, disease), the vegetation in this area remains intact or recovers remarkably quickly whilst other surrounding areas are decimated. In marine ecosystems, for example, it has been noted that some areas of coral do not undergo coral bleaching despite the surrounding coral being decimated by this event (e.g. Penin et al., 2012). Factors responsible for areas of resilience are complex and, as yet, poorly known. However, it is widely recognized that some landscapes, because of their composition of abiotic and biotic features, appear to have the ability to withstand intervals of climatic disturbance, disease, or some other environmental catastrophe. These landscapes are important to any ecosystem because they provide the refuge for biodiversity during environmental perturbations and enable sustainability of populations through time. Resilient areas are therefore a key ecological feature that should be conserved on any landscape.

4.2.3 Spatial resolution of datasets

When aiming to protect biodiversity, how much land needs to be conserved? Spatial scale is probably where conservationists and businesses/habitat managers are furthest apart. For the biodiversity conservationist, the aim is usually to consider (and conserve) an area as large as possible to encompass as many habitats and ecosystems as possible. Biologically this makes sense; especially if considering ecological functions as well as properties; river valleys, intact habitat patches, and other such features can cover hundreds of kilometres of landscape. The spatial scale for modelling and prediction of patterns of diversity across space and species responses to climate change, etc. are equally coarse, and are normally mapped at the scale for which the best regional climatic models are available, which is currently at a resolution of about 0.5°. This in effect means that there is one biodiversity data point every 25–50km.

In contrast, for the business/habitat manager, where the focus is on the landscape (outside of a protected area) in terms of its conversion to other uses, the aim is to find the smallest area that needs to be conserved. The exact location and spatial scale also tends to be tightly constrained by legislation. The requirement of the business/habitat manager is therefore to understand the relative biodiversity within a granted concession area, and at as small a scale as possible (usually at a resolution < 0.5km), a scale of analyses at a much finer spatial resolution than that normally considered for biodiversity conservation and placement of protected areas. The measures also need to be relative within that landscape; how does one area compare with another in terms of its relative ecological properties and features? Admittedly, some metrics, such as threatened species, will rely upon knowledge of the global distribution of a species to assess their threat status. However, beyond protected areas, the overall focus will be on the number of threatened species in one place (i.e. pixel) relative to another in the same landscape (i.e. which area carries the highest ecological risk if destroyed).

4.2.4 Temporal resolution of datasets

Temporal datasets are particularly important for assessing two key biodiversity features: recovery rates and resilience. Understanding recovery rates is important because it has been demonstrated many times that even across landscapes of a few kilometres there is wide variation in time taken for a damaged habitat to recover. Factors responsible for faster recovery rates can be due to the internal dynamics of the system—in the Canadian boreal forest, for example, it has been demonstrated that soil type (determined by geology) strongly influences recovery rate; so too does vegetation type, with black spruce demonstrating a particularly slow recovery rate (Lee and Boutin, 2006). The black spruce forest therefore carries a much higher ecological risk if damaged.

Determination of both recovery rates and resilience therefore requires datasets that illustrate the variability of the biota (plant or animal) through time. In terms of length of dataset, the longer the better because, given that the average generation times of most trees and large organisms is >50 years, then datasets shorter than this often do not cover one full generation of the main organism under consideration (Willis et al., 2010). Such temporal datasets do exist in historical records (photos and documents), and reconstruction of past variability using fossil records is possible, but in the absence of these there are now satellite imagery records spanning > 30 years that can provide a preliminary record of response to environmental perturbations that have existed in the past decade.

4.3 WHAT INFORMATION DO WE ALREADY HAVE?

There are now a number of good datasets and schemes available to identify some of the most biodiverse regions globally. These range from global measures that use a variety of metrics to identify whole landscapes important for biodiversity through to local presence of individual species. Metrics to determine spatial patterns of biodiversity include the 'hotspot' approach, where important regions for biodiversity are identified based on a measure of the number of species/endemics and threat that a region is under (e.g. through land-use change) (Myers et al., 2000; Brooks et al., 2006), through to those that identify geographically distinct assemblages of natural communities (e.g. WWF Ecoregions[4]). To date, thirty-four biodiversity hotspots have been identified globally, 825 terrestrial ecoregions, 426 freshwater ecoregions, and 229 coast and shelf marine ecoregions.[5] In addition, there are areas recognized for their importance to particular species, such as Important Bird Areas,[6] or the habitat that they provide for that species in terms of a migratory route.

Often in conjunction with various schemes to identify landscapes most in need of protection, there are also a number of schemes that have been developed to determine the global distribution of species most under threat of extinction. The IUCN 2010 Red List of Threatened Species, for example, contains assessments of ~58,000 threatened species globally, and spatial distribution maps of ~28,000 of these species, along with a measure of their status ranging from 'extinct in the wild' through to 'vulnerable', and then 'of least concern'.[7] Movement across landscapes is also mapped globally for a number of species. The global registry of migratory species (<http://www.groms.de>) currently contains a list of 2,880 migratory species in digital format and digital global migration maps, detailing important landscapes/routes for 545 vertebrate species. Another landscape-scale scheme is the UNESCO Man and Biosphere programme. There are 560 biosphere reserves in over 100 countries which have been identified as important because of their combination of both biotic and cultural diversity.[8] There are also key biodiversity areas (KBAs) that identify regions containing overlapping distribution ranges of species with high conservation priority.[9] A similar landscape-scale approach has now also been developed to determine important marine biodiversity hotspots (Roberts et al., 2002), and this has led to the creation of marine protected areas.[10]

[4] <http://worldwildlife.org/biomes>.
[5] <http://www.worldwildlife.org/science/ecoregions/item1847.html>.
[6] <http://www.birdlife.org/action/science/sites/>.
[7] <http://www.iucnredlist.org/>.
[8] <http://www.unesco.org>.
[9] <https://cmsdata.iucn.org/downloads/pag_015.pdf>.
[10] <http://www.wdpa-marine.org/#/countries/about>.

When these datasets are viewed globally they indicate an impressive distribution of protected areas.[11] A number of excellent web-based landscape strategic conservation planning tools have also been developed for further determination of conservation priorities within these protected landscapes (Aitken et al., 2008; Pressey et al., 2009): Marxan (Ball et al., 2009) and Zonation (Moilanen, 2007), including those that select the best reserves for conservation based on the uniqueness of a reserve and its overall contribution to biodiversity targets (Carwardine et al., 2007). In essence, these tools provide a means to rank the importance of reserves, so that sites with rarer biodiversity features will have a higher irreplaceability value than sites with more common features. Measures of irreplaceability are then used to determine which reserves to prioritize for action. Two of the most widely used 'irreplaceability tools' are C-Plan (NPWS, 1999)[12] and Marxan.[13] Datasets that are used to determine irreplaceability include species composition data, vegetation types, rarity, etc. With C-Plan analysis it is also possible to include resource data such as timber yields. A site with a high conservation value and low timber production, for example, would indicate an important region to conserve because not only is it a good option for biodiversity conservation, but it will also have minimum impact on timber resources.

In determination of the location of protected areas and their spatial location at landscape scale, it would therefore appear that the knowledge base is fair. Where the datasets and tools to model biodiversity become much less impressive, however, is in the determination of biodiverse landscapes beyond protected areas (i.e. 87 per cent of the Earth's terrestrial surface). There is also a strong spatial skew in the availability of data. Some countries (e.g. USA, Canada, Australia, Finland, and New Zealand) have excellent high-resolution datasets of species occurrence, rare species, endemics, and other biotic and abiotic features that enable remote mapping of biodiversity to a high level of detail and at a fine spatial scale (1–10km resolution). NatureServe, for example, is a web-portal that provides information and access to databases detailing occurrence data on more than 50,000 plants, animals, and ecological communities of the USA and Canada (<http://www.natureserve.org>). For many countries, however, in-country data-portals are non-existent, and to obtain information, a combination of global datasets, models, and algorithms needs to be utilized. An overview of some of these methods is presented in the following section.

4.3.1 Richness

Richness can be seen as a result of the combination of the total species diversity in a given place (habitat level—*alpha diversity*) and/or the diversity of habitats

[11] <http://www.protectedplanet.net/>.
[12] <http://www.uq.edu.au/ecology/index.html?page=101951>.
[13] <http://www.uq.edu.au/marxan/>.

(*beta diversity*) (Whittaker, 1972). In an ideal world, there would be sufficient data to obtain a picture of these two variables through assessing biodiversity on the basis of individual species occurrence data combined with species distribution models (e.g. Hirzel and Guisan, 2002). For most regions in the world, however, this is not possible due to lack of species occurrence data. An alternative strategy in such situations is to shift the focus from individual species to emergent properties of biodiversity (Ferrier et al., 2002). Such an approach links the limited information available from species occurrence data to a suite of environmental variables to make statistical inferences about biodiversity in locations about which we know environmental characteristics but do not have species information.

This approach has been successfully tested for both alpha and beta diversity using a suite of modelling approaches. Leathwick and colleagues (Leathwick et al., 1998), for example, applied General Additive Models to predict alpha tree diversity in New Zealand. In this study, alpha-diverse sites in New Zealand's primary forests were preferentially found in regions with high temperatures, high solar radiation, soil and atmospheric moisture, and on sedimentary and basaltic substrates. These habitat factors were then used to predict similar habitats that would be suitable for such species elsewhere. A similar approach can also be used to calculate beta diversity (diversity of habitat). Ferrier and colleagues, for example (Ferrier et al., 2002; Ferrier et al., 2007) have developed a Generalized Dissimilarity Model that determines beta diversity based on extant species data and environmental covariates. This model can then be used to predict similar patterns of beta diversity elsewhere; an approach has been extensively used in many environments (Ferrier et al., 2002; Ferrier et al., 2007; Willis, et al., 2012; Blois et al., 2013).

In order to determine either alpha or beta diversity across global landscapes (i.e. outside of protected areas), there is still the issue of where the data to compute these factors are going to be obtained. However, there are now a number of excellent global databases that can start to provide this information remotely. Most notable is the Global Biodiversity Information Facility (GBIF).[14] This is a data repository and portal for all geo-tagged species data. GBIF currently contains records of 380 million species occurrences worldwide. For environmental data there are several sources for spatially explicit environmental data (i.e. climate, soil properties, distance to water), including the Worldclim Climate Grids, the Harmonized World Soil Database, the Hydrosheds drainage channels, and Global Lakes and wetlands databases.[15]

[14] <http://data.gbif.org>.

[15] Respectively <http://www.worldclim.org>; <http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML>; <http://hydrosheds.cr.usgs.gov/>; and <http://worldwildlife.org/pages/global-lakes-and-wetlands-database>.

4.3.2 Vulnerability

Vulnerability is most often determined as the number of endangered and/or threatened species occurring over a given area. One database commonly used to undertake such assessments is the 2010 IUCN Red List of Threatened Species, which provides geo-located species occurrence data and spatial range maps of endangered species at a coarse spatial scale made by expert opinion. In order to overcome the spatial coarseness and the lack of spatial data for many species in the IUCN, species distribution models have also been applied to many endangered species for which there is observational data (Guisan et al., 2006).

A vast ecological literature exists that demonstrates the ecological importance of habitat integrity and the impact of fragmentation on biodiversity (see Fahrig, 2003 for a review). In general, the greater the patch size, the higher its functionality (greater diversity, more pollinators, greater complexity in food webs, etc.), and thus the greater its ecological value (Willis et al., 2012). A key feature to try and retain in any landscape, therefore, is habitat integrity. A good approach to calculating ecological patch size is to determine similar adjacent vegetation types. One way to do this is to use the GLOBCOVER dataset which delivers global land cover maps at a spatial resolution of 300m.[16] Existing software can be then used to define patch size distribution (e.g. FRAGSTATS, McGarigal and Cushman, 2002) to determine the size of similar vegetation cover.

As already mentioned, measuring connectivity is critical to the study of fragmented populations (Prugh et al., 2008), since it is essential to any ecologically functioning landscape. Connectivity across a landscape is usually achieved through river corridors and/or other migratory routes. Information on river corridors and/or wetlands is available globally—e.g. the HYDROSHEDS river network database (Lehner et al., 2008),[17] and the Global Lakes and Wetlands database (Lehner and Döll, 2004). In addition, the Global Register of Migratory Species GROMS (<http://www.groms.de>) (see earlier) can be used to calculate the number of migratory species ranges which intersect a given target area (e.g. Willis et al., 2012).

4.3.3 Resilience

The ability to measure resilience of landscapes in response to environmental perturbations is currently limited by the availability of spatially complete

[16] <http://due.esrin.esa.int/globcover>.
[17] <http://worldwildlife.org/pages/hydrosheds>.

long-term ecological and environmental records. Nevertheless, some studies to determine resilience of landscapes to environmental perturbations have been attempted with some success for certain areas. For example, Klein et al. (2009), working in the arid interior of Australia, determined which parts of the landscape were most resilient to drought stress. They did this by acquiring values of annualized gross primary productivity from high-resolution time-series satellite data, as well as time series of climatological data, and identified the places with the highest productivity during the least productive years over a five-year period from July 2000 to June 2005. Using this approach it was therefore possible to identify regions that maintained maximum productivity despite minimum precipitation. Similar approaches have been implemented in Willis et al. (2012).

In many other regions of the world plant productivity will most likely be limited by a mix of many environmental variables. In order to implement the approach by Klein et al. (2009) in a global framework, therefore, the variables controlling primary productivity have first to be identified. Such an approach holds great potential for providing a global picture of resilient sites in space and time. Moreover, the identification of particular years which have caused non-linear responses (i.e. outlier years) in plant productivity should serve to identify environmental conditions which have led the local vegetation to cross potential thresholds (Willis et al., in preparation).

4.3.4 Future distribution of biodiversity

In light of current accelerated climate and land-use changes, the ecological value of any site is affected not only by its present biodiversity outcome, but also by its future likely biodiversity outcome. Ideally, a detailed knowledge of every species' physiological response to climate change would be needed to this end. However, this approach requires huge amounts of data, and information of this nature is available only for a small number of species. An alternative is again the use of statistical species distribution models. These take the present-day distribution of the species and the climatic envelope in which it is found, and this is used to create a 'species envelope', or 'bioclimatic envelope' model, which calculates a relationship between the species and its associated climatic parameters. If the model is then run with future climate data it can be used to demonstrate the potential change in climate-space for the species/communities under a different climatic scenario.

Such an approach has been adopted for hundreds of species for which there is observational information to predict future distributional shifts and threats to biodiversity (e.g. Thuiller et al., 2005). However, the sensitivity of these models to the use of different algorithms and spatial resolutions, as well as the uncertainties related to responses of biota to increased carbon dioxide concentration (carbon dioxide fertilization effect) and biological interactions (i.e. changes in the competition intensity between species due to changes in climate) have been addressed but not fully resolved (e.g. Araújo et al., 2005; Araújo and Luoto, 2007; Rickebusch et al., 2008; Randin et al., 2009). Promising results have recently come from the independent validation of species distribution models using palaeoecological data (e.g. fossil pollen and remains of plants and animals, including ancient DNA). This approach has increased confidence in predictions stemming from these models (e.g. Macias-Fauria and Willis, 2013), as well as expanding knowledge about species tolerance to environmental conditions not observed today (e.g. Martínez-Meyer et al., 2004; Blois et al., 2013) and potentially relevant in the future. Likewise, the integration of key components of mechanistic species models (e.g. plant response to increased atmospheric carbon dioxide) into statistical models offers great potential (Kearney and Porter, 2009). This highlights the potential of species distribution models to assess future safe and unsafe areas for biodiversity, and scenario-setting for future ecosystem service provision.

In summary, even in the absence of high-resolution species occurrence data, there is an increasing number of models, algorithms, and datasets that can be used to obtain a first approximation of current species distributions and important habitats for biodiversity. In addition, methods to predict future distributions and areas of resilience in response to climatic perturbations are now coming on-line and have great potential for landscape planning at local, regional, and global scales.

4.4 TOOLS AVAILABLE TO ILLUSTRATE SPATIAL PATTERNS OF BIODIVERSITY AND ECOSYSTEM SERVICES

Whilst the models, algorithms, and datasets described earlier can provide a means of measuring different aspects of biodiversity beyond protected areas, the next hurdle is the collation of the information into a format that is appropriate for end-users (policy-makers, governments, businesses). Over the past decade a number of web-based tools have been developed that coordinate some of these methods and databases to aid stakeholders in making decisions about activities which may have consequences for biodiversity and ecosystem service provision. These include handbooks and checklists, procedures for ranking, algorithms for economic valuation, and mapping tools. Probably the most useful for landscape planning, however, are those that generate an output map of the pattern of biodiversity and/or ecosystem services in relation to the activities of a business.

The main web-based mapping tools currently available fall into two categories: those that display a particular feature(s) of the landscape (e.g. water, threatened species, protected areas), and those that attempt to provide a synthesis of various features to create an assessment of the land for the ecosystem services that it contains. In the former category, for example, is the Global Water Tool, which focuses on evaluating likely impacts of business operations on water supplies by comparing the location with the best available water, sanitation, population, and biodiversity information on a country and watershed basis.[18] Biodiversity is assessed in terms of the location of the site in relation to Conservation International's Biodiversity Hotspots (see earlier)—i.e. closeness of site to protected areas. This is probably fair given that the key focus of the tool is climate/water/population. However, consideration of other aspects of the watershed (and probably some information that is already available in the raw data) could provide much more in terms of assessment of the ecological risks associated with the development. These include, for example, assessment of the land cover and potential fragmentation due to the business development, and consideration of how the impact of the development will damage the ecosystem services provided—e.g. clean water filtration and soil erosion.

The Normative Biodiversity Metric (NBM) is another web-based tool that attempts to provide a measure of a single feature—this time the 'pristineness' of the land on which the proposed development is to take place on a 0–5 scale, where 5 is pristine, 4 is minimal use, 3 is impacted, 2 is converted, 1 is monoculture, and 0 is artificial.[19] To calculate the Biodiversity Metric, two databases are accessed—namely the WWF Terrestrial Ecoregions and Ellis and Ramankutty's (2007) World Map of Anthropogenic Biomes (Ellis and Ramankutty, 2007). However, both are highly derived products, neither use primary field data in the analyses, and they provide output at a coarse resolution (10–50km). Considerable downscaling has therefore taken place with this metric in order to provide output in the tool at a local landscape resolution.

A tool that attempts to assess the biodiversity of the landscape using a number of different features is the Integrated Biodiversity Assessment Tool (IBAT).[20] This web-based tool integrates site location information supplied by businesses with data from global databases on the location of protected areas, key biodiversity areas, Alliance for Zero Extinction sites, global range polygons of threatened species on the IUCN Red List, and maps of broad-scale conservation priorities such as Biodiversity Hotspots and Endemic Bird Areas. The overarching aim of this tool is to help businesses to understand the global and regional biodiversity context of the sites and identify risks associated with their

[18] <http://www.wbcsd.org/work-program/sector-projects/water/global-water-tool.aspx>.
[19] <http://nbm.ourecosystem.com/interface>.
[20] <http://www.ibatforbusiness.org/login>.

own site location. In this respect it is a valuable tool because it collates information from a number of biodiversity databases and highlights the proximity of the site to key protected sites and threatened species across global landscapes. It is also easy to use and the outputs (maps) are easy to obtain. Where it is less successful, however, is in its spatial coverage (normally >5km pixel resolution) and also its focus on protected areas and species. In terms of addressing and/or mapping important areas for biodiversity outside of protected areas (both static features and those important to maintain ecological processes, as discussed previously), such a tool is unable to address the question 'where can we damage?'

Also in the category of determining a number of biodiversity/ecosystem features across the landscape of interest is the Local Ecological Footprinting Tool (LEFT).[21] This web-based tool was originally developed to aid industry in evaluating the pattern of relative ecological value across a landscape to inform their planning of land use in order to minimize the environmental impact of their operations. A user defines an area of interest anywhere globally using a web-based map and the tool then automatically processes a series of high-quality datasets and databases (described earlier) using standard published algorithms (Willis et al., 2012) to produce maps at 300m resolution of land cover class, numbers of globally threatened terrestrial vertebrate and plant species, beta diversity of terrestrial vertebrates and plants, habitat fragmentation, wetland habitat connectivity, numbers of migratory species, and vegetation resilience. These results are also aggregated to produce a single map of relative ecological value. The tool then generates a customized pdf report and a zip file of geographic information system (GIS) data for the area requested.

What becomes apparent from briefly reviewing some of the main tools currently available for assessing landscape patterns of biodiversity is how few of them (with the exception of IBAT and LEFT) access the excellent resources provided by the numerous global databases identified in section 4.3. In addition, most rely upon species occurrence data that were specifically collated for protected areas and are therefore of limited use outside of protected areas. The consideration of biodiversity in most cases is static and calculated on the here-and-now (and simple metrics) rather than using the suite of peer-reviewed models that are now available to obtain a dynamic picture of biodiversity and/or broader ecological considerations including connectivity, fragmentation, and resilience. The advantage of all of these tools in comparison with the next set of tools, however, is their ease of use; they all have a simple web user-interface and require minimal (if any) input from the user.

[21] <http://www.biodiversity.ox.ac.uk/left>.

There are five tools that are currently recommended for determining ecosystem service provision across landscapes: (i) the Artificial Intelligence for Ecosystem Services (ARIES);[22] (ii) Co$ting Nature;[23] (iii) EcoMetrix;[24] (iv) Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST);[25] and (v) Multi-Scale Integrated Models of Ecosystem Services (MIMES).[26]

However, all of these ecosystem service tools (with the exception of Co$ting Nature) require a high level of technical (often GIS) expertise to extract the necessary information, a large amount of user-supplied data is required to run the models, and skilled GIS technical input with estimations of between 20 and 200 hours to complete one assessment leading to the following conclusion in a 2011 assessment on the state of biodiversity markets:[27]

> a gap in biodiversity market infrastructure that persists is lack of landscape-scale ecological monitoring. While site-level ecological monitoring is not uncommon, the data is not easily available, much less compiled in a comprehensive way. (Madsen et al., 2011)

4.5 KNOWLEDGE GAPS

So far this chapter has focused on existing approaches and challenges for measuring biodiversity and mapping ecosystem services in human-dominated landscapes. This final section focuses on four interconnected challenges, and associated knowledge gaps that emerge when trying to decide which landscape components of biodiversity need to be conserved.

First, despite our growing understanding of how different human activities can affect biodiversity, the fact remains that this knowledge is fragmented and at best incomplete for a number of activities across the world. During project planning, potential impacts on biodiversity are often established through proxies such as expected land-use change or pollution. For example, species-area models have often been used to predict or infer biodiversity loss from land-use change due to human activity. However, the results of such predictive exercises are often highly uncertain as important background knowledge is missing. To complicate matters further, often the same human activity can have radically different biodiversity impacts depending on management decisions related to location and infrastructure. It is therefore imperative to understand the biodiversity impacts of such management practices if biodiversity is to be effectively conserved beyond protected areas. Tools such as

22 <http://www.ariesonline.org>. 23 <http://www.policysupport.org/costingnature>.
24 <http://www.parametrix.com>. 25 <http://www.naturalcapitalproject.org>.
26 <http://www.afordablefutures.com/services/mimes>.
27 <http://www.bsr.org/reports/BSR_EMI_Tools_Application1.pdf>.

LEFT and IBAT can provide a first approximation of the biodiversity across landscapes, but the challenge now is to develop similar tools/data layers that can determine the potential ecological footprint of different human activities at a similar landscape scale of resolution so that the two can be mapped alongside each other. Given the large scope of this endeavour and the limited amount of resources available, cost-effective techniques need to be sought to obtain this information, including, for example, greater use of new automated technologies for gathering data using smartphones and tablets, and the involvement of citizen-scientists (Snaddon et al., 2013).

Second, most ecosystem services studies seem to focus on a limited number of provisioning services (e.g. food, fibre, fodder, water provision) and regulating services (e.g. climate regulation). Understanding the flow of these services provides only a piecemeal understanding of the benefits that can be derived from a landscape, and this represents a significant knowledge gap. More information is needed on how other services, such as pollination, erosion regulation, and the numerous supporting and cultural services, are provided by *in situ* landscape elements, and how these services can be linked to biodiversity. Furthermore, there have been only a few studies to examine if (and how) landscapes provide multiple ecosystem services (bundles), and if these ecosystem services bundles are spatially correlated (Raudsepp-Hearne et al., 2010). Integrated studies are increasingly required if we are to better understand the complexities of ecosystem services trade-offs.

Third, there is a lack of data on how ecosystem services are linked to human well-being. Only when these links are fully appreciated can the true costs and benefits of biodiversity conservation beyond protected areas be calculated. Extensive meta-analyses of published literature can be a first step to establishing such links for different human activities and landscapes across the world (e.g. Roe and Elliott, 2010). Such meta-analyses can be ideal for identifying specific knowledge gaps and for forming hypotheses about the nature of these mechanisms. Substantial empirical research will then be needed to establish such links and quantify human well-being impacts. This will require approaches that can combine insights from the natural and the social sciences and integrate meaningfully qualitative and quantitative data. Multi-criteria-based techniques hold significant promise but have not been used extensively in ecosystem services studies so far.[28]

Fourth, there is a lack of appropriate integrated assessment tools. As already mentioned there is a large gap in the availability of tools that can be used to assess multiple ecosystem services in a rapid and robust manner. Current ecosystem services assessment tools often require huge amounts of detailed

[28] <http://www.teebweb.org>.

data input and specialized knowledge (see the previous section). This can prove to be an important barrier to their adoption by the private sector. As an extension it can hinder conservation efforts beyond protected areas, considering that such areas are, to a large extent, privately owned. An important research task for the future therefore is to develop integrated assessment mechanisms fit to assess such trade-offs at the landscape level. Such tools need to be able to provide a robust and integrated assessment of the many impacts associated with the diverse human activities. They should also be able to consider different scenarios. Only through such comparisons will it be possible to identify the least damaging development strategies in a given landscape.

4.6 CONCLUSIONS

'We need more data' is probably the one sentence that policy-makers and managers most dislike hearing from scientists. So does this sentence hold true when considering the opening question of this chapter—i.e. where can we damage? More data is certainly highly desirable, especially for some of the most biodiverse regions in the world—but it is also clear that there is already a vast amount of data and resources available to address this question. Given that some of the global databases outlined in this chapter, such as GBIF, now contain over 380 million species occurrence data points, and that other equally impressive data collations are available for threatened species (e.g. IUCN Red List of Threatened Species), and location of protected areas (e.g. WDPA), there needs to come a point where pragmatism kicks in and we work with what is already available.

Taking as a starting point the assumption that there is enough data to make meaningful observations, the next question is whether we are using this data to its full potential to address the questions of where can we damage, and if not, why not? It is in these questions that the examples presented in this chapter start to highlight some areas for improvement.

First, very little use is currently made of existing databases such as GBIF for assessing patterns of biodiversity. It is particularly notable, for example, that of the different web-based tools for assessing biodiversity only one (LEFT) uses GBIF data, while there are literally only a handful of papers published in the past year that cite GBIF as the data source. Second, the vast majority of research effort for identifying biodiversity is still restricted to protected areas and usually with a focus on species decline. This needs to change quickly. Third, with two exceptions (LEFT and IBAT), the suite of algorithms and models that have been devised to assess biodiversity beyond protected areas within web-based tools are extremely simplistic and in some cases not fit-for-purpose. A suite of features needs to be addressed, as outlined, for example, by de Groot et al. (2010), in order to determine the biodiversity potential and risk of landscapes including both static elements (e.g. current biodiversity patterns) and dynamics features (e.g. connectivity across the landscape, fragmentation, etc.). Most assessments currently look at only one or two of these features to make assessments. Fourth, when it comes to assessing ecosystem service flows from landscapes beyond protected areas the opposite is true—at least in terms of modelling. Very advanced models have been developed, but these require a large amount of time and an advanced level of knowledge from the user, relying extensively on user input to obtain a meaningful output. In addition, biodiversity data is rarely considered within these models/tools.

These problems are research challenges rather than impenetrable barriers. They have almost certainly emerged because of a lack of dialogue between natural/social scientists and stakeholders to determine the key questions and pragmatic solutions to biodiversity conservation beyond protected areas, as well as a lack of knowledge of the available data/models. Now is the time to work directly with the landowners and businesses to ask the difficult questions, because without their input and buy-in, the future for global biodiversity looks bleak.

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5

The UK National Ecosystem Assessment: Valuing Changes in Ecosystem Services[1]

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5.1 INTRODUCTION

The UK National Ecosystem Assessment (UK NEA), commissioned by the Department for Environment, Food and Rural Affairs (Defra) and published in June 2011, involved more than 600 scientists from a wide range of disciplines. The primary tasks of the UK NEA were to assess the state of the UK's ecosystems and the services provided by them, how they have evolved over the last fifty years, and how they are likely to change in the coming fifty years. The scenario analysis[2] we summarize here is part of the economic assessment conducted within the UK NEA. Despite being a relatively small part of the overall exercise it nevertheless sparked considerable interest in the media and had a substantial impact on UK environmental policy. The purpose of the scenario analysis was to provide a consistent and highly comparable assessment of changes in key ecosystem services—including biodiversity—for a set of six plausible scenarios for the year 2060. It allows a synthesized but at the

[1] This work was funded in large part by the *Social and Environmental Economic Research (SEER) into Multi-Objective Land Use Decision Making* project (which in turn is funded by the Economic and Social Research Council (ESRC); Funder Ref: RES-060-25-0063). Further funding was provided by the UK National Ecosystem Assessment (UK NEA). This chapter is based on Chapter 26 of UK NEA (2011).

[2] A more detailed description can be found in Chapter 26 of the UK NEA Technical Report (2011) while more recent analysis is presented in Bateman et al. (2013).

same time spatially explicit assessment of these changes in benefits derived from ecosystem services. It demonstrates that the necessary tools exist to conduct impact assessments that go well beyond narrow market values to include monetary valuations of ecosystem services such as recreation, greenhouse gas emissions, and urban greenspace amenity. Two indicators of biodiversity are included, but not monetized due to concerns regarding the robustness of available monetary measures. The analysis therefore presents ways in which biodiversity can be integrated into such an assessment under such circumstances.

The purpose of economic analysis is to aid decision-making which seeks to examine the trade-offs implied by each of a set of feasible options, so identifying that option which offers the best net benefits for society. For this reason, economic analysis is less interested in the total value of ecosystem services (not least because for essential services total values may be infinite) than in the change in value generated under one state as opposed to another. The scenario analysis therefore assesses moves from a common baseline to each of the states described under the NEA scenarios, and considers the changes they imply for selected ecosystem services and the value of those changes.

We do not pretend to value the impact of future scenarios on all ecosystem services. This is in part a reflection of the state of available data and knowledge. Economic values (for any good, not just ecosystem services) are contextual—i.e. marginal values (the value of a single unit change in a good) vary across space and time. So, for example, the value of a recreational visit may vary according to a variety of locational factors. This information is not available for all of the ecosystem services. Because of this the scenario analysis focuses on a subset of ecosystem-service-related goods for which we do have sufficient data to undertake defensible valuations. Obviously this subset does not represent the totality of values generated in the move from one state to another. Consequently, the valuations reported are necessarily partial and provisional and should not be taken as indicating the overall value of ecosystem service changes arising under each scenario. Nevertheless, for the first time they bring together both key market and non-market benefits of land-use changes at the national level in a way that allows them to be directly compared and aggregated.

Despite these caveats, the scenario analysis of the UK NEA amply demonstrates that methods now exist to unite natural sciences with economic assessments so as to estimate the value of changes arising under different states and thereby inform decision analysis. This is, arguably, the most important finding of the NEA in terms of its implications for the future. It paves the way for a new approach to decision-making in which ecosystem services can be directly incorporated into policy choice.

5.2 VALUING ECOSYSTEM SERVICES

In this section we briefly outline the methodologies used in the scenario valuation of the UK NEA. First, the six NEA scenarios are introduced. We then turn to each of the five ecosystem services included to conduct a series of highly comparable scenario analyses. The ecosystem service goods considered are:

- Agricultural food production;
- Terrestrial carbon storage and greenhouse gas (GHG) emissions;
- Biodiversity (assessed using birds as an indicator species);
- Open-access recreation;
- Urban greenspace amenity.

5.2.1 The NEA scenarios

The six NEA scenarios are briefly outlined in what follows. For a more detailed description see Chapter 25 of the UK NEA (2011). The scenarios were derived by considering the impacts of changes in both the regulatory setting and climate change as these are predicted to be the main drivers of land-use change over the next decades. In contrast, factors such as soil types are likely to remain constant over the time horizon considered here.

Each scenario makes specific assumptions about the stringency of future environmental regulation and planning policy relative to the current one, and some scenarios focus changes on particular areas such as peri-urban ones or those of particular conservational value. These assumptions are then translated into land-use changes at the level of 1km squares.

- ***Go with the Flow (GF)***—essentially follows today's socio-political and economic trends. It presents a future that is roughly based on today's ideals, with some leaning towards improving the environmental and sustainability performance of the UK. Environmental improvements are still important in the government's vision for a future UK, but the public is less keen on adopting many global or national environmental standards (business and industry even less so).
- ***Green and Pleasant Land (GPL)*** is a storyline where the conservation of biodiversity and landscape are the dominant driving forces. Whilst it is recognized that biodiversity often provides essential benefits to society, its intrinsic value is accorded a pre-eminence in policy and legislation. The countryside is very much a managed, cultural landscape, but the focus is now on maintaining and improving the aesthetic appeal. In general, landscape preservation often coincides with biodiversity conservation.

- ***Local Stewardship (LS)*** has localism as a dominant paradigm. It is also more environmentally aware and open to international trade than some other scenarios (e.g. *National Security*, see later). Political power has been devolved and many major issues are decided at a regional or local level. GDP is low but sustainable. People travel less and depend more on local resources; more food production and leisure activities take place in the immediate locale. The sustainable management of resources is a priority, and society relies less on technological innovation. Low-carbon economies and alternative economies such as LETS (Local Exchange Trading Systems) schemes are more common. Through local specialization the UK becomes less homogenized—landscapes become more distinct. Social and environmental regulation has advanced.
- Under the ***National Security (NS)*** scenario UK industry is protected from foreign investors and imports. Trade barriers and tariffs are increased to protect jobs and livelihoods in the UK; immigration is also very tightly controlled. Technological development is state-funded and many industries (including agriculture) are subsidized by the state. Food, fuel, timber, and mineral resources are prioritized over biodiversity conservation.
- In the ***Nature at Work (NW)*** scenario the conservation of biodiversity as an end in itself is less of a priority compared with maintaining and enhancing the output of ecosystem services. Adapting to climate change is also a priority, which means that some non-native species are introduced to provide food, energy, or shade. Promoting ecosystem services in multi-functional landscapes is now embedded in all walks of society. Habitat restoration and creation are seen as important components of this campaign, but the explicit conservation of species is sometimes overruled by a 'greater' ecosystem service benefit. Modern technology is used where appropriate though, and even GM biotechnology is adopted if it can be shown to enhance ecosystem service provision. 'Optimal Service Provision' is key, and many ecosystem services in the landscape are the result of careful examination of the trade-offs through scientific and community review.
- In the ***World Markets (WM)*** storyline unfettered economic growth through the complete liberalization of trade is the main goal. International trade barriers dissolve, agriculture subsidies disappear, and farming, for example, is now industrial and large-scale. Consumption in society is high, which results in greater resource use and imports. There is competition for land, and this, coupled with reduced rural and urban planning regulations on housing, agriculture, and industry, means that biodiversity is often the loser. Fish stocks plummet and a few species have been wiped out. Supplies of other ecosystem services increasingly become privatized.

All of these scenarios were further modified according to two different responses to climate change, as taken from the UKCIP-09 Low and High Emissions Scenarios for 2050–79.[3] Here we focus on the high emission versions. The full set of scenarios is discussed in Chapter 26 of the UK NEA.

For each scenario, changes in the value of ecosystem services provided are calculated between a baseline (2000) and the envisioned state of the UK in 2060 under the respective NEA scenario. The valuation of changes under each scenario informs of the trade-offs across the set of goods under consideration.

An alternative approach to the scenarios used here is to analyse the impacts of specific policy proposals. This route is taken in the ongoing second phase of the UK NEA.

5.2.2 Agricultural food production

The agricultural section of the analysis compares agricultural land types and livestock numbers under the baseline with those in each scenario. Based on this we derive the economic impact on farmers in terms of Farm Gross Margin (FGM), defined as the difference between revenues from agricultural activities and associated variable costs. Hence, neither fixed costs nor conversion costs are included.

First, specific land uses and livestock numbers for the farmland areas (for each 2km grid square) in both the baseline and all NEA scenarios are predicted using the CSERGE (Centre for Social and Economic Research on the Global Environment) econometric agricultural land-use model (Fezzi and Bateman, 2010; Fezzi et al., 2010*a*). This ensures that results are consistent with the behavioural patterns observed throughout its large cross-sectional and time-series database. Based on estimated land uses we generate the corresponding FGM values, which are then contrasted with the baseline to estimate the change in value induced under each scenario (full details are presented in Fezzi et al., 2011).

The agricultural analysis is highly spatially explicit, allowing decision-makers to target policies at those areas where they generate the most efficient use of resources. National-level estimates of the values of changes induced under each scenario are detailed in Table 5.1. Here, the upper row details the baseline, highlighting the significant heterogeneity which characterizes the British farming system (for example, the FGM/ha of the third quartile is more than seven times that of the first quartile).

Achieving higher environmental quality (the GPL and NW scenarios) would come at some cost for the farming community (overall between 1 per cent

[3] UKCIP is the UK Climate Impacts Programme hosted at the Environmental Change Institute, University of Oxford.

Table 5.1. Summary statistics for FGM per hectare in the 2000 baseline and in the various 2060 scenarios (real values, £, 2010)

Scenario	Mean	Lower quartile	Median	Upper quartile	Total GB	Δ GB	Δ GB
	£/ha	£/ha	£/ha	£/ha	£m pa	£m pa	%
Baseline	173.1	34.9	223.4	268.6	3,100		
GF High	205.9	34.8	227.4	301.3	3,690	590	*19.0*
GPL High	171.5	34.8	198.0	254.8	3,070	−30	*−1.0*
LS High	196.8	33.3	223.8	299.7	3,530	430	*13.9*
NS High	239.8	25.3	269.2	340.1	4,300	1,200	*38.7*
NW High	167.0	31.5	164.8	253.3	2,990	−110	*−3.5*
WM High	222.2	38.9	242.3	308.9	3,980	880	*28.4*

Note: FGM taken from Fezzi et al. (2010*b*) as follows: cereals = £290/ha, root crops = £2,425/ha, oilseed rape = £310/ha, dairy = £576/head, beef = £69/head, sheep = £9.3/head. Δ is change in total value compared with the baseline. Due to data availability these results apply to Great Britain (GB) rather than the UK.

Source: UK NEA Technical Report (2011).

and 10 per cent of total FGM for GPL, and between 4 per cent and 20 per cent for NW). However, here the distributional impact of these losses is progressive with poorer farmers being relatively unaffected (the first-quartile income does not change), while incomes amongst richer farms decline noticeably (note the fall in third-quartile incomes).

Encouraging agricultural production under the NS and WM scenarios will, as one would expect, boost agricultural incomes. However, the total amount of agricultural land decreases significantly under these scenarios, depressing aggregate gains. In particular, the scenarios envisage a loss of low-productivity rough grazing and permanent grassland. However, the overall value of agricultural output is expected to increase.

5.2.3 Terrestrial carbon storage and GHG emissions[4]

The changes in annual GHG emissions from terrestrial ecosystems resulting from changes in land use and associated land management draw directly on the CSERGE land-use model as reported in the preceding section. They therefore share the same methodology and assumptions in determining both agricultural land use and livestock numbers. Both of these are important determinants of the GHG balance. For example, land use influences carbon storage, while methane and N_2O emissions from grazing livestock represent important

[4] This section draws on Abson et al. (2011).

Table 5.2. Change in the value from baseline year of annual GHG emissions from GB terrestrial ecosystems in 2060 under the high-emission variants of the NEA scenarios (£ million/yr); negative values represent increases in annual costs of GHG emissions

GF	GPL	LS	NS	NW	WM
−812	2,410	567	3,393	4,569	−1,675

Source: UK NEA Technical Report (2011).

sources of terrestrial GHGs. Information on changes in woodland extent was taken directly from the NEA scenarios.

Three major categories of GHG emissions were considered in estimating changes in annual GHG emission flows:

- Emissions from land use and land management: (1) due to energy use from farmland activities such as tillage, sowing, spraying, harvesting and the production, storage and transport of fertilizers and pesticides; (2) emissions of N_2O and methane from livestock; (3) N_2O emissions from artificial fertilizers.
- GHG emissions/accumulations from land-use change: for example, permanent grassland converted from arable farming will be accumulating soil organic carbon (SOC), while permanent grassland previously under rough grazing may be losing SOC.
- Emissions/accumulations of carbon in terrestrial vegetative biomass.

For the baseline year annual GHG emissions from terrestrial ecosystems are estimated to be 26 $MtCO_2e$ (million tonnes of carbon dioxide equivalent). Land-use management represents the dominant source of emissions in the baseline.

The UK government's official non-traded marginal abatement cost of carbon (MACC) prices (DECC, 2009) are used to value the changes in annual emissions from 2000 to 2060 under each scenario. This means that carbon prices are set at £41.28 per tCO_2e (tonne of carbon dioxide equivalent) in 2000, and are increasing to £273.50 per tCO_2e in 2060. Table 5.2 shows the change in the annual costs of GHG emissions from GB terrestrial ecosystems compared with the baseline year for each scenario. This means that positive (negative) values represent an increase (decrease) in costs. Three of the scenarios (GPL, NS, and NW) show significant reductions in annual costs associated with emissions of GHG.

5.2.4 Biodiversity

While there is a variety of methods available for monetizing the use value of biodiversity, monetary estimates of its non-use existence value can be obtained only via stated-preference methods. While a number of such studies have been undertaken, critics question whether the values estimated for such a low-experience good as biodiversity are based on the robust preferences required for admission within cost–benefit analyses (CBA). While we do not pass judgement on this matter, we demonstrate that, even in the absence of monetary estimates of non-use existence values, there are useful inputs which economic analyses can provide to decision-makers. In particular, economists can advise on the cost-effectiveness by comparing the levels of both biodiversity and other economic values arising in different NEA scenarios. This approach reveals the costs of improving biodiversity. We term this a 'cost-effectiveness analysis' (CEA). Clearly this is not as desirable a state as knowing the monetary value of that biodiversity and entering it within a CBA; nevertheless, at least the feasible trade-off is now explicit. Furthermore, win–win situations might exist where both biodiversity and the monetary value of other goods increase. In the present section we quantify the biodiversity impacts of each scenario. In the final section of this chapter we then compare the valuation of all monetized outcomes of each scenario with those impacts.

This section uses birds as indicators of biodiversity. Birds are a prominent aspect of UK biodiversity, are high in the food chain, and are often considered to be good indicators of wider ecosystem health (see, for example, Gregory et al., 2005). Birds are more mobile than most other groups, so will respond to, and reflect, environmental quality on a rather broader scale than mammals or terrestrial insects. This makes them better indicators at the landscape level but less good locally. However, we do not suggest that they provide a comprehensive summary of all aspects of biodiversity. Rather, we note the value that birds have as indicators and make use of the important pragmatic benefit that they are better monitored than any other aspect of UK biodiversity. Our first analysis takes a wide view across almost all (here: 96) GB bird species, while the second analysis focuses on farmland birds as the group that has suffered the most dramatic declines over the past half-century. In both cases measures of bird success are modelled as a function of land use. These models are then used to assess the predicted impact on these bird measures as a result of the differing land uses envisioned under each of the NEA scenarios, again drawing on the land-use model presented previously.

Breeding bird diversity as a function of land cover

The model used for this analysis is discussed in detail by Hulme and Siriwardena (2010). Essentially it links GB data collected by the British Trust for Ornithology/Joint Nature Conservation Committee/Royal Society for the Protection of Birds Breeding Bird Survey (BBS) with land-use information provided by the CEH Land Cover Map 2000. The composition of the bird community represented by the presence and abundance of bird species in each survey square was summarized using Simpson's Diversity Index (D), which is given by the inverse of the sum over the squared species proportions.

The mean value of D was calculated for each square across all years within the study period in which that square was surveyed, and this was modelled alongside the habitat and land-use classes from the Centre for Ecology and Hydrology (CEH) Land Cover Map 2000. Models were run predicting diversity at a 1km^2 level. Diversity was then predicted for each of the NEA scenarios.

In general, the GF, GPL, and NW scenarios all lead to some modest increase in bird diversity in lowland areas. While this might be as expected for the overtly pro-environmental GPL and NW scenarios, the increase in diversity under the GF scenario indicates that this is set against the ongoing commitment across society to biodiversity-friendly management—for example, as reflected by the Common Agricultural Policy 'Pillar 2' investment in agri-environment measures, as well as a general leaning towards biodiversity under this scenario. However, all three of these scenarios also reveal a slightly more pronounced decrease in diversity in upland areas as climate change induces increases in relative agricultural intensity within these areas. This trend is broadly reversed for the LS and NS scenarios, and becomes most extreme under the WM scenario, although here we also see some declines in upland areas. Indeed, across all scenarios, it is the WM case which gives both the greatest declines (–0.131) and largest increases (0.040) in predicted bird diversity.

The patterns of change predicted under each of these scenarios are summarized in Table 5.3. All changes in absolute diversity values are well below 10 per cent. Thus, the predictions indicate minor changes in bird communities, rather than local extinctions or colonizations. The present analysis provides an indication of the type of quantitative biodiversity analysis which can be set alongside economic benefit valuations within a cost-effectiveness analysis.

Table 5.3. Summary statistics showing the predicted changes in bird diversity from the 2000 baseline to NEA high-emission scenarios for 2060

Scenario	Mean	Lower quartile	Median	Upper quartile
GF	0.00175	0.00000	0.00118	0.00336
GPL	0.00467	0.00000	0.00372	0.00879
LS	−0.00024	−0.00203	0.00015	0.00195
NS	0.00870	0.00022	0.00327	0.01522
NW	0.00396	0.00000	0.00243	0.00659
WM	−0.00434	−0.00735	−0.00087	0.00139

Note: All statistics are summaries across all 235,974 1km squares in Great Britain for which mapped predictions were available, and so represent the average changes across the whole country and the variability in these patterns. Mean standard error <0.00005 in all cases.

Source: UK NEA Technical Report (2011).

Table 5.4. Summary statistics for the change in guild richness for 19 species of farmland birds from the baseline to 2060 under each of the NEA high-emission scenarios

	GF	GPL	LS	NS	NW	WM
Mean	−0.42	−0.37	−0.39	−0.84	−0.62	−0.47
Mean (% change)	−2.2	−1.9	−2.1	−4.4	−3.3	−2.5
Lower quartile	−1.89	−1.85	−1.85	−2.26	−2.10	−1.91
Median	−0.48	−0.47	−0.49	−0.85	−0.73	−0.58
Upper quartile	0.95	0.97	0.95	0.61	0.72	0.87

Source: UK NEA Technical Report (2011).

Habitat association modelling for farmland birds

Changes in farming practices contributed to a 52 per cent decrease in the England farmland bird index between 1970 and 2009 (Defra, 2010). These bird species are important not only as indicators of wider biodiversity, but also in their own right.

The model used for this analysis is discussed in detail by Dugdale (2010). It considers a single 'guild'[5] of nineteen, primarily farmland, bird species. Guild richness was measured as the number of these species present in each 10km grid square in England and Wales. Models were developed linking guild richness to data on land use, woodland, and urban extent.

Results of the scenario analysis are summarized in Table 5.4. The mean impact of all scenarios is a reduction in guild richness, although this is generally not large enough to generate a one-species change in typical 10km grid squares. Nevertheless, four scenarios reduce mean guild richness by more than 0.5 (NS High, NS Low, NW High, and NW Low), suggesting that, on average, one species fewer would be present under these scenarios.

As suggested in Table 5.4, there is considerable variation in predicted guild richness across scenarios.

5.2.5 Open-access recreation[6]

The valuation of open-access recreation involves three linked analyses:

(i) A site-prediction model (SPM) predicts the number and location of recreation sites under each scenario. Sites are predicted using data from the Monitor of the Engagement with the Natural Environment (MENE) provided by Natural England, Defra, and the Forestry

[5] Defined as a group in terms of the common foods they consume—here primarily seeds and invertebrates.

[6] This section is based on Sen et al. (2011).

Commission to model the relationship between site location, land use and the proximity to, and density of, population.

(ii) A trip-generation function (TGF) predicts the number of day visits from any outset location to any specified site as a function of the availability of substitutes around the outset location, the population of that outset area and their socioeconomic and demographic characteristics, and the physical environmental characteristics of destination sites.

(iii) A meta-analysis of estimates of the value of visits, taking into account the nature of any visited site.

By combining outputs from the SPM and TGF, we predict both where sites will be and how many day visitors they will attract. By feeding this estimate into the meta-analysis model we obtain an estimate of the value of those visits. This yields estimates of recreational value which are sensitive to the spatial distribution of populations and their characteristics, and the spatial distribution of recreational sites and their environmental characteristics. This in turn ensures that the methodology is sensitive to the populations and land-use changes envisaged under the NEA scenarios.

The distribution of sites and visits shows the estimated total number of visits to each grid cell per annum. This distribution conforms strongly to prior expectations. Visit numbers reflect the very strong influence of travel time as well as of the land-use and habitat types of each area. For example, prized landscapes such as large areas of south-west England, the north Norfolk coast, the western coast of Wales, and the border areas of Scotland down into the Lakes all exert a pull on visitors which overcomes the fact that they have relatively low resident populations.

The total annual visitor numbers can then be fed into the meta-analysis model to convert visitor numbers into values, taking into account the land-use and habitat characteristics of each visited site and their corresponding specific values. Repeating this exercise for the baseline and all NEA scenarios generates the values presented in aggregated form in Table 5.5.

In general, the value changes are dominated by increases in visit value. The NW scenario displays the most substantial increases in the value of visits for large areas of Great Britain at both high and low emissions. In most scenarios, large impacts are seen in and around urban areas, while more rural areas are

Table 5.5. Changes in total value (£ million) of predicted annual visits under the various scenarios (high-emissions versions)

GF	GPL	LS	NS	NW	WM
4,121	5,156	1,098	3,344	2,3914	−823

Source: UK NEA Technical Report (2011).

affected less. Larger predicted reductions are seen under the LS scenarios, particularly in the area south and west of London and in the urban centres, although London itself shows a substantial increase in the value of visits. The WM scenario probably shows the greatest difference in comparison with the other scenarios. There, London shows a very large decrease in value of visits, with similar decreases in predicted visit value also seen in other urban centres across the country. In all cases the remote uplands, because of their inaccessibility, remain unvisited and show no change in value.

At the national level all of the scenarios generate increases in the annual value of visits except for the WM scenario (Table 5.5). In general, we find large gains under the NW, GPL, and GF scenarios and moderate increases for the LS scenario.

5.2.6 Urban greenspace amenity

Key ecosystem services provided by urban greenspace in the UK include recreation, aesthetics, physical and mental health, neighbourhood development, noise regulation, and air pollution reduction. They are provided as a bundled good and should be valued as such. We undertake a meta-analysis of prior studies allowing us to estimate how amenity values decline with increasing distance between households and urban greenspace areas. Capturing this distance dependence is vital if we are to accurately assess the value of changes in the number, extent, and location of urban greenspaces as cities and their populations alter in the NEA scenarios.

The six NEA scenarios detail a number of changes to key urban characteristics such as their physical extent, their population, and the area of urban greenspace provided. Table 5.6 presents the percentage changes for these key variables.

The implicit changes in greenspace access (and hence distance decay in values) were assessed through geographical information system (GIS) analysis of distance relationships for a set of five UK urban centres (ranging from relatively small cities like Norwich to major conurbations like Glasgow). This analysis provided information on the proximity of each household to urban greenspaces, both under the 2000 baseline and for each of the NEA scenarios.

Perino et al. (2011) combine the GIS data with the value functions derived from the meta-analysis to calculate values for the changes in urban greenspace for both the set of cities considered and the implied values for the whole of Great Britain;[7] it is these latter, national-level values that we focus on here. In calculating these, value estimates are made only for cities with a population

[7] Comparable data for Northern Ireland is not available. However, urban areas in Northern Ireland represent only about 3 per cent of total urban area in the UK (see Chapter 10 of the UK NEA).

Table 5.6. Changes in urban characteristics from the 2000 baseline to 2060 for each of the NEA scenarios

Scenario	Change in urban area (%)	Change in urban population (%)	Change in the area of formal recreational space (%)	Change in the area of informal greenspace (%)
GF	3.0	32.2	36.2	0.0
GPL	0.0	21.7	38.9	5.4
LS	−3.0	0.0	4.5	2.8
NS	−3.0	17.2	−34.3	4.8
NW	−3.0	13.8	39.0	−4.9
WM	79.0	52.6	73.0	20.7

Source: UK NEA Technical Report (2011).

Table 5.7. Per-household and aggregated benefit changes of scenarios for Great Britain

	GF	GPL	LS	NS	NW	WM
	Per-household (£)*					
Undiscounted value change	−7,800	9,300	8,500	−39,300	18,700	−94,700
Annuity (infinite, 3.5%)	−114	136	125	−576	274	−1,390

Note: * Based on the 15.2 million urban households living in the areas included in the extrapolation.
Source: UK NEA Technical Report (2011).

of 50,000 or more as the methodology used is regarded as less suitable for smaller settlements.

The set of sampled cities allows us to calculate the value of changes in urban greenspace for more than 1,600 urban areas (defined as Census lower super output areas (LSOA) in England, and Census data zones in Scotland). Regression analysis linked these value estimates to a variety of small area characteristics. Given that these predictors of value can be obtained for all Census areas of all cities, the model can be used to extrapolate value changes across Great Britain. Table 5.7 presents the resulting valuations of the changes in urban greenspace envisioned under each scenario.

While these values should be regarded only as approximations, they underline the very substantial changes in urban greenspace values which can arise across these scenarios. While more extreme scenarios such as World Markets lead to very substantial losses in urban greenspace values, even moderate scenarios show that feasible changes to urban greenspace can generate significant changes in values.

The changes in amenity value provided by urban greenspace are driven by a combination of factors. A change in the size of a city changes the average distance to nearby greenspace and hence the amount of benefits (e.g. recreation, cleaner air, aesthetics, etc.) realized by urban households. An increase in urban population, *ceteris paribus*, decreases per-household benefits as parks become increasingly crowded.

While under constant pressure due to the increasing demand for housing and commercial development, urban greenspace generates substantial benefits to local communities. This analysis shows that changes in the provision of urban greenspace can create, or destroy, billions of pounds worth of benefits to local residents.

5.3 SYNTHESIS OF ECOSYSTEM SERVICE VALUATIONS

The scenario analysis of the UK NEA has considered five ecosystem service goods: agricultural food production, terrestrial carbon storage and annual GHG emissions, biodiversity (assessed using birds as an indicator species), open-access recreation, and urban greenspace amenity. For each of these goods we have examined the changes in provision between a baseline set as the situation in 2000 and the envisioned state of the UK in 2060 under the NEA scenarios. With the exception of biodiversity, all of the goods are valued in money terms.

Here we synthesize the results for the five ecosystem services at the aggregate, national level. Great care has to be exercised in the interpretation of such synthesis findings. Most obviously, while this analysis goes beyond the normal decision remit of purely market values, it considers only a small subset of ecosystem-service-related goods. Many market and non-market values are omitted here and so the analysis is necessarily partial and incomplete. Similarly, we are not considering the extent to which different scenarios impinge upon international trade and the effective import of ecosystem services (e.g. water embodied in agricultural imports) and resultant export of an ecological footprint. While these are important caveats, they do not undermine the fundamental objective of this analysis, which is to demonstrate that methods for the integrated valuation of highly varied goods have now been developed. However, there is an obvious danger of a simplistic acceptance of the following results as representing the value of all changes induced under any scenario. This would be highly erroneous and must be resisted. Nevertheless, what this demonstration does illustrate is that methods exist which address many of the key challenges to the incorporation of ecosystem services and the wider values of the natural environment within practical decision-making. Furthermore, even this partial analysis amply shows that such incorporation can radically alter the apparent value of a given scenario or policy option. As such, these techniques point to a superior basis for future decision-making.

Table 5.8 summarizes results from the various analyses outlined earlier. It is important to recall that all of the values and impacts recorded here relate to changes rather than totals. So, for example, the agricultural values reported are simply for the change in value relative to the baseline. It is the change in value induced by policy or other drivers which should be the focus of decision analysis allowing an informed choice between options.

Examining the monetary values reported in Table 5.8 reveals a number of interesting findings. A general observation is that the magnitude of value changes within the farm provisioning services is generally lower than those of non-marketed goods. This is immediately important as it is only agricultural values which are reflected in market prices. Analyses such as those provided by the NEA are vital if we are to ensure efficient decision-making and an optimal allocation of resources. The last four rows of Table 5.8 underscore this message. The first of these ranks the NEA scenarios solely according to the market value they generate; here represented by agricultural produce. We can see that most of the scenarios generate improvements in market (agricultural) values relative to the present day, particularly the

Table 5.8. Summary impacts for the change from the 2000 baseline to 2060 under each of the high-emission NEA scenarios: Great Britain (£ million per annum)

	GF	GPL	LS	NS	NW	WM
	£m pa (real values, 2010)					
Market agricultural output values*	590	−30	430	1,200	−110	880
Non-market GHG emissions†	−810	2,410	570	3,400	4,570	−1,680
Non-market recreation‡	4,120	5,160	1,100	3,340	23,910	−820
Non-market urban greenspace§	−1,960	2,350	2,160	−9,940	4,730	−24,000
Total monetized values‖	1,940	9,890	4,260	−2,000	33,100	−25,620
	Non-monetized impacts					
Change in farmland bird species	0	0	0	−1	−1	0
Bird diversity (all species)#	++	++	–	++	++	–
Rank: market values only	3	5	4	1	6	2
Rank: all monetary values	4	2	3	5	1	6
Rank: +ve monetary values and no farmland bird losses	3	1	2			
Rank: +ve monetary values and biodiversity gains	2	1				

Notes: * Change in total GB farm gross margin.

† Change from baseline year (2000) in annual costs of GHG emissions from GB terrestrial ecosystems in 2060 under the NEA scenarios (£m/yr); negative values represent increases in annual costs of GHG emissions.

‡ Annual value change for all of Great Britain.

§ Annuity value; negative values indicate losses of urban greenspace amenity value.

‖ We acknowledge some double counting between urban recreation and urban greenspace amenity values. Further data is needed to correct for this.

\# Based on relative diversity scores for all species.

Source: Abstracted from Bateman et al. (forthcoming) and UK NEA Technical Report (2011).

National Security (NS) and World Markets (WM) scenarios. Conversely the Green and Pleasant Land (GPL) and Nature at Work (NW) scenarios yield losses (indicated by the grey shading) when assessed in market value terms alone. The subsequent row extends our analysis to include all monetized values, irrespective of whether they are generated in markets or not. Here the ranking of scenarios changes dramatically, with the NW scenario moving from being the worst to now being the best option in terms of social value, and the GPL coming second to this. In a similar manner the NS and WM scenarios, which were ranked as best in terms of market values alone, now appear to yield the two worst outcomes in terms of their overall social value. This is a major message of the NEA: omission of non-market values can result in socially sub-optimal situations, or even outcomes which actually reduce overall social welfare.

The final two rows of the table progressively exclude scenarios purely on the basis of their biodiversity outcomes. The penultimate row ranks outcomes only for those scenarios which both generate net social benefits and which avoid any further losses to our priority farmland bird diversity measure. This leads to the rejection of the NW scenario, because its reduction in agricultural area results in localized losses of some farmland bird species. However, the opportunity costs of rejecting this scenario (which actually improves other biodiversity measures) are substantial, amounting to a loss of net social benefits of over £20,000 million per annum, or two-thirds of the net value of the NW scenario. The final row of Table 5.8 further restricts the analysis to only those scenarios which deliver biodiversity gains, although in this application the optimal scenario does not alter.

5.4 POLICY IMPACT AND CONCLUSION

The main objective of the UK NEA was to assess the past and future changes of the UK ecosystems and the services provided by them. The scenario analysis summarized here shows that an integrated economic analysis covering changes in both key market and non-market benefits is now feasible. An important contribution of this work is to provide such a highly integrated assessment at different spatial scales, reaching from 1km squares in non-urban and full postcodes in urban areas to the national level. The methods and marginal value function can be used to assess policies at any of these scales, allowing the construction of toolkits that can substantially improve decision-making.

Some caveats are in order. The set of ecosystem services covered by this analysis is far from complete. The services provided by water and the risks it poses (e.g. by flooding) are missing completely. Biodiversity is represented only in the form of proxies (here: two measures of bird diversity) and not valued in monetary terms. In urban areas, currently only the amenity values of urban greenspace are captured, while arguably there are other relevant services provided as well. So clearly, the UK NEA is only a first step towards an encompassing, integrated, spatially explicit assessment of ecosystem services—albeit an important one. A critical appraisal of the valuation methods used can be found in Chapter 6 of this book (Atkinson et al.).

The UK National Ecosystem Assessment had a significant impact on environmental policy in the UK. In June 2011, just a week after publication of the UK NEA, Defra published its Natural Environment White Paper, 'The Natural Choice: Securing the Value of Nature', mapping out environmental policy for the years to come and using the NEA as its main evidence base, citing it more than fifty times. Key policies contained in the White Paper, such as the Natural Capital Committee, Nature Improvement Areas, and the Green Infrastructure Partnership, are directly based on findings of the NEA, with explicit reference being made to the results of the scenario analysis.

The newly created, independent Natural Capital Committee reports to the Economic Affairs Committee (which is chaired by the Chancellor of the Exchequer), and advises on the state of natural capital in England and how changes in stocks of natural capital can be integrated into national economic accounting. This provides a direct and long-term influence on the valuation of ecosystem services of the type conducted by the UK NEA to feed directly into economic policy at the highest level.

The Green Infrastructure Partnership, led by Defra, brings together government agencies, stakeholders, and academics. It combines expertise, transfers knowledge on greenspace valuation to decision-makers, and promotes the consideration of non-market benefits of greenspaces in planning decisions.

Following the successful completion of the UK NEA, a second phase of work was announced by the UK Secretary of State for the Environment at the 'Planet Under Pressure' conference in London during March 2012. This follow-on phase broadens and deepens the research started by the UK NEA. In particular, the work described in this chapter is extended to discriminate between more land-use types, with a focus on the effects of afforestation being of special concern. A further extension is to develop new models of the consequences of land-use change for the water environment, initially in terms of water quality, but longer-term to include quantity issues such as flooding and droughts. A more fundamental change is to move away from a scenario-driven approach with its focus on end-states, and move instead towards a drivers approach, where changes in the policy, market, technological, and environmental determinants of land use are followed through to examine their consequences both in terms of land use and for the integrated systems considered. Other aspects of the UK NEA follow-on work being tackled by parallel teams include a focus on the development of tools to incorporate research findings such as these within decision-making systems. Together, this research provides a useful perspective on the integration of natural science, economics, policy, and decision-making.

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Part II

Valuing Biodiversity

6

Valuing Ecosystem Services and Biodiversity

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6.1 INTRODUCTION

There are arguably few more obvious manifestations of interest in the economics of ecosystems and biodiversity than the prominence now given to economic valuation. Broadly speaking, what this activity refers to is placing monetary values on the many services that are provided (in large part) by the natural world and consumed by human populations. It is important to ask: what is the question (or what are the questions) to which knowing these values could be the answer? One response would be the growing recognition that the benefits and opportunity costs associated with such services are frequently given cursory consideration in policy analyses, or even completely ignored. The valuation of biodiversity and ecosystem services is therefore a crucial element of robust decision-making. In this respect, this valuation is a means of extending practical cost–benefit thinking to the domain of nature conservation. Related motivations have led to use of valuation in large-scale evaluations of ecosystems, such as the UK Natural Ecosystem Assessment and possible ambitious extensions to comprehensive wealth accounting.

The critical starting point for this work is the recognition that only rarely do the services of biodiversity and ecosystems command a market price. This is not to say that these values are not reflected at all in market prices. Nature is after all critical to sustaining significant portions of economic activity. The point here is that this contribution is likely to be reflected only indirectly in the market prices of the goods and services that are traded as a result of this activity. Moreover, nature could well provide services which increase human well-being but are not reflected in markets, whether this be indirect or otherwise. Within environmental economics, uncovering these values is the domain of non-market valuation.

From humble origins in the early post-war period (e.g. Hotelling, 1949), the literature regarding the valuation of people's preferences for non-market costs and benefits has grown, initially slowly, but more recently at an almost exponential rate. Categories of application abound. Significant attention has focused on valuing human health and longevity (see, for a review, Pearce et al., 2006). These applications have transformed the way in which cost–benefit appraisals of, for example, transport proposals and air-quality-management strategies are conducted. In this context, valuing biodiversity and ecosystems is an extension of this work-in-progress to a vitally important further policy area. However, this extension should not be viewed as merely routine. It poses challenges that are at the frontier of non-market valuation and, indeed, may even hint at the limitations of these methods.

Perhaps most fundamental is the need to ensure that such applications are based on a sound foundation of natural science. Indeed there is a highly cogent case to be made that all such applications necessarily require interdisciplinary collaboration between, at a minimum, the natural sciences and economics (arguably extending to a much wider fusion of disciplines). This requirement for interdisciplinarity is given a conceptual framework within the so-called 'ecosystem service' approach to decision-making. While typically characterized as emanating from the natural sciences, the approach is highly compatible with economic analysis as it emphasizes the role of ecosystems in providing services which, in turn, either support production or are direct contributors to well-being. Ecosystem services are therefore defined as contributors to anthropocentric values, and while the natural sciences provide an understanding of the former, it is economics which is well placed to assess the latter. Economic valuation, in particular, becomes an essential element of the ecosystem service approach to decision analysis.

While the term 'ecosystem services' is relatively recent, being popularized only in the wake of the Millennium Ecosystem Assessment (MA, 2005), environmental economists have been applying non-market valuation techniques to such services for many years (see, for example, Adamowicz et al., 1994; Ruitenbeek, 1989). Understanding the economic value of ecosystems and biodiversity is important for a number of reasons. One of these is undoubtedly the perceived persuasiveness of economic language. That is, conveying what it is that the natural world provides us with in monetary terms is seen as a powerful means of communicating the importance of conservation to a wider (and perhaps previously unreceptive) audience. For example, Bateman et al. (2011*b*) estimate that, in the UK, ecosystem services help contribute to 3 billion outdoor recreational visits annually, with the social value of the output created by these trips likely to be more than £10 billion. Gallai et al. (2009) calculated the global value of the services provided by insect pollinators to be about $190 billion (in 2005) just in terms of the benefits arising from pollination of crops for (direct) human consumption.

But beneath the rhetoric there is genuine substance in that these data can also be used to guide policy-thinking and decisions. In the case, for example, of the recreational value of UK ecosystems, Bateman et al. (2011*b*) also show how location (of these sites) matters. A specific and moderately sized nature recreation site, for example, might generate values of between £1,000 and £65,000 per annum depending solely on where it is located. The critical determinant of this range is perhaps not surprisingly proximity to significant conurbations. Put another way, woodlands in the 'right' place (i.e. relatively close to potential visiting populations) are likely to give rise to higher social values (other things being equal), an insight of particular importance if policy-makers are contemplating new investments in these nature sites.

More generally, the key insight in explicitly placing a value on nature is that it redresses a fundamental imbalance whereby this value is—all too frequently—grossly misjudged or just plain ignored in private and (much of) social decision-making. And while debates about the intrinsic value of nature remain relevant, demonstrating that nature has significant instrumental value for human livelihoods or human well-being more broadly is then a crucial practical step in developing policy actions that address current and projected rates of ecosystem destruction and biodiversity loss. One much cited example in this respect is Barbier (2007). That study estimates the ecological value of mangroves in Thailand—in terms of providing fuelwood, a habitat that supplies fisheries and storm-water attenuation (which reduces the risks of coastal flooding)—in order to compare those findings with the returns from the competing land-use activity of shrimp farming. Thus, private profits under these two different uses are $584 and $1,220 per hectare respectively, giving, on the face of it, a clear (financial) case for mangrove conversion. However, social cost–benefit analysis reveals another story in that a representative hectare of mangrove is shown to generate a social value of $12,392.

These benefits that nature provides might even spill over to human populations living in countries other than where, say, an ecosystem is sited. In a study of Costa Rica's tropical forests, Bulte et al. (2002) conclude that the optimal area of forest land is more than twice as large as the actual (1998) area once the value of domestic externalities provided by this forest is taken into account. Bringing the value of global externalities (accruing to those outside of the country but provided by Costa Rica's forests) into this reckoning results in the optimal forest cover being calculated to increase by a further 20 per cent. Of course, the economic approach may not always provide us with the answer that ecosystems or biodiversity should be protected (and thus indicates the pitfall for those who see only the rhetorical worth in economic arguments). Nevertheless, and however the question is posed, determining how much of nature needs to be conserved is likely to require a significant effort to understand its value in economic terms as well as the (opportunity) costs of its conservation.

Any chapter that seeks now to take stock of efforts to value ecosystem services and biodiversity has the advantage of following a number of comprehensive reviews such as Kumar (2010), Bateman et al. (2011*b*), ten Brink (2011), and specific reviews of, for example, forests and coastal/marine ecosystems (see, respectively, Ferraro et al., 2012 and Barbier, 2012). In what follows, while we inevitably will draw on these important contributions, we also hope to add to insights about past and future endeavour in this field. In the section that immediately follows we briefly review possible classifications of ecosystem services, but discuss in addition the more recent—but hugely important—development that traces further links to the underlying ecological assets that give rise to these services in the first place as well as the role of biodiversity. Section 6.3 outlines the key valuation methods and considers, in particular, gaps in the empirical record and the scope for filling these gaps. Section 6.4 sets this consideration of economic valuation in relation to the evidence base needed to inform broader ecosystem assessments and policy decisions. Section 6.5 concludes.

6.2 A FRAMEWORK FOR VALUING ECOSYSTEM SERVICES AND BIODIVERSITY

In the past few years, interest in the problem of ecosystem and biodiversity decline has grown dramatically, among academics and policy-makers alike. Much of this recent attention can be traced to the MA (2005), which made clear the scale of the challenge at hand in its identification of persistent and growing threats to ecosystems around the world. In addition, the focal valuation message in the Stern Review on Climate Change (Stern, 2007) appears not to have been lost on decision-makers within the domain of conservation policy. Assessments including the G8/EU-initiated TEEB Review (The Economics of Ecosystems and Biodiversity, TEEB, 2010) and the UK National Ecosystem Assessment (UK NEA, 2011) can be viewed as an attempt to generate a correspondingly increased awareness and strong policy response for biodiversity and ecosystem services, as well as a concerted effort to build on the momentum and insights generated by the MA.

Importantly, the MA had the effect of broadening the focus of concern from just biodiversity loss to cover, in addition, the loss of ecosystem services, with the critical emphasis of the latter on 'the benefits people obtain from ecosystems' (MA, 2005, p. 53). From an economic perspective, ecosystem services are simply those contributions of the natural world which generate goods which people value. The term 'goods' is, as elsewhere in environmental economics, construed widely to mean physical products and less tangible outputs. This includes services which generate use values, and non-use goods which are valued purely for their continued existence.

This now conventional understanding (within environmental economics) of the total economic value of some 'good' has been intertwined with a more nuanced understanding of the specific services that ecosystems provide. There are a number of variations on these classifications. Common to almost all is a distinction between provisioning services, cultural services, and regulating services. The former two services nicely capture some elements of the previous distinction between use and non-use. Provisioning services, for example, are typically physical products such as food and natural materials provided by nature. Cultural services, by contrast, describe the experiences that people enjoy as a result of interactions with nature (e.g. recreation) as well as more intangible pleasures arising from knowledge about the existence of nature or its spiritual value.

Further classifications of ecosystem services do exist. Kumar (2010), for example, adds habitat services in recognition of the role that ecosystems provide in protecting 'gene pools' as well as crucial sets of interlinking habitats for migratory species. MA (2005) also emphasizes the supporting services of ecosystems as the natural processes that underpin those services of provision, culture, and regulation. These services, such as nutrient cycling, thus provide a further intermediate tier to ecological production and, indeed, it is has since become more common to see these functions subsumed under the 'regulating services' heading (e.g. Kumar, 2010). Other classifications such as Heal et al. (2005) and De Groot et al. (2002) have focused more specifically on habitat services and regulating services. While this emphasis is partial it encapsulates a key distinctive element of the effort to understand the economics of ecosystems. This likens the enjoyment of (final) ecosystem services to a process of (natural) production whereby critical inputs are, for example, regulating services. As an illustration, it is these services—by, for example, regulating water flow (and the quality of that water) and the supply of insect pollinators—that contribute ultimately to the production of agricultural provisioning services (Goulder and Kennedy, 2011). Valuing ecosystem services has often focused on the end output by asking what is the final service that ultimately benefits people. Clearly, knowledge of what ecosystems provide as final goods and services that we consume is important. Yet it is equally crucial that we understand the way in which intermediate tiers of production contribute to this final output.

In many ecosystem classifications (including those which have been expanded to conceptualize ecosystems as assets), there appears to be no explicit place for the value of *biodiversity*. Indeed, a significant anxiety about recent ecosystem assessments is that the emphasis on ecosystem services might ironically lead to the omission of the vital role which biodiversity plays in both the delivery of those services and as a source of value in itself. Mace et al.

(2012) provide clarification of the issue, noting that biodiversity appears at three distinct points within the ecosystem service framework.

First, as discussed in detail by Elmqvist and Maltby (2010), biodiversity acts as a supporting service underpinning the delivery of what Fisher et al. (2009) term final ecosystem services. So, for example, soil biodiversity enhances farmland fertility, which in turn determines production of a good (here food). In fact, such functions provided by biodiversity have been likened by, for example, Pascual et al. (2010) to a form of insurance (following from earlier contributions such as Gren et al., 1994). According to this view, a more diverse (ecosystem asset) portfolio has a distinct value in terms of maintaining resilience: that is, the capacity of a system to persist, in some state, in the face of shocks and stresses that it might experience (Perrings, 2006; Mäler et al., 2009).

Second, biodiversity acts as a final ecosystem service itself. For example, pollinator biodiversity directly enhances agricultural production. Third, certain aspects of biodiversity, such as the continued existence of iconic species such as the polar bear, themselves constitute a good (i.e. a direct source of well-being). These diverse roles suggest that attempts to value biodiversity will be challenging. It is to these challenges, and those entailed in valuing ecosystem services, to which we now turn.

As reflected in our discussion thus far, much of the existing terminology in ecosystem valuation and biodiversity conservation has focused on *services*: that is, some flow of a benefit arising perhaps from the consumption of a good or broader amenity. Of course, policy interventions such as investments in ecosystem protection (or enhancement) typically will boost the flow of these services over time, thereby introducing a dynamic element into any economic analysis. Moreover, when ecosystems are perturbed by some change (be it a shift in land use or a degradation in state), the effect on well-being similarly will have an intertemporal dimension (e.g. Mäler, 2008; Dasgupta, 2009). Put this way, what we need to think about is the underlying ecosystem asset and, in particular, the changes in asset value that occur as a result of human interventions (be these positive or negative, deliberate or otherwise). Broadly speaking, what needs to be assessed here is the potential change in our *future* prospects given what is happening to ecosystems now. Thinking about ecosystems as assets (as opposed to emphasizing only current services) is in its relative infancy, but is becoming more prominent. In the view of Heal (2007), this brings the study of the economics of the natural world in line with other areas of the discipline. Barbier (2009) has shown how this extension of the ecosystems analytical framework results in a more explicit conceptual understanding of ecosystems as complex assets giving rise to multi-dimensional services.

Thinking explicitly about ecosystems as assets thus opens up a further range of valuation issues. That is, given that a change in asset value is equal to the difference in the present value of future services before and after the change, we need to consider how these future services are to be valued and, moreover, discounted. Clearly, neither of the implied measurement challenges is unique to valuing ecosystem assets. Questions about asset valuation (as well as answers to those questions) pervade many other areas of economics. Ongoing efforts to measure ecosystem and biological assets can usefully learn much from these existing insights. For example, the debate that ensued after the Stern Review (e.g. Dietz and Stern, 2008; Weitzman, 2007) has thrown new light on the choice of the *social* discount rate in the context of climate change. A recent review by Gowdy et al. (2010) in the context of ecosystems and biodiversity illustrates that the issues are likely to be no less controversial given the long-term characteristic of services provided by nature. However, to date this has received far less attention in this context (see, for example, Mäler et al., 2009, for a brief discussion in the context of ecosystem accounting).

Discussion of ecosystems and biodiversity has also focused on the ability of valuation methods, for practical purposes, to deliver on addressing concerns about the complexity of ecosystems and the empirical relationship between asset stocks, the flow of services, and the way in which these services are valued at different stock levels (Pascual et al., 2010). This is a point that can be traced back at least as far as Krutilla and Fisher (1974), but has been made more recently, and often with ecological wealth in mind, for the case of assets for which there are limited substitution possibilities (in terms of the well-being that they ultimately provide). That is, the (marginal) value of the service (i.e. its relative price) is likely to increase all the more rapidly as the asset is increasingly degraded or converted.

Gerlagh and van der Zwaan (2002), for example, consider the case where these substitution possibilities are a function of the asset stock itself. That is, when a resource such as an ecosystem is relatively abundant, losses in that asset 'do not matter' in the sense that this source of well-being could be easily replaced with something else and people essentially would be no worse off. However, above some thresholds, substitution possibilities diminish rapidly. In other words, continued loss of the natural asset—beyond a particular critical threshold—increasingly cannot be compensated and instead increases the prospect of significantly raised adverse impacts on future well-being.[1] Hoel and Sterner (2007) and Sterner and Persson (2008) have indicated some initial steps towards a practical exposition of this thinking (in the context of valuing the damage arising from climate change). However, this empirical progress requires that a number of assumptions must be made: most notably, a

[1] Gerlagh and van der Zwaan (2002) look at the case where individuals have a very strong preference for natural assets rather than non-substitutability per se. This is very similar to the notion of a lexicographic preference that has been the subject of a mini-literature in stated-preference studies. The implications of this assumption, however, are that liquidating a natural asset beyond some threshold plausibly lowers the maximum level that future well-being can take.

judgement needs to be arrived at about the 'elasticity of substitution' (between some natural asset and other productive stocks). Further investigation of these issues, within the ecosystem context, is urgently needed (although see Barbier, 2009, for a discussion on modelling the likelihood of collapse of ecosystem assets, and Farley, 2008, on the broad principles that might guide future thinking about valuation as ecosystem assets become increasingly scarce and, in some cases, stocks approach critical levels).

Finally, it is also worth noting that one particular approach to thinking about ecosystems as assets addresses a possibly critical issue with regard to diversity (and discussed earlier) by treating 'ecological resilience' as a stock (Mäler et al., 2009, Mäler, 2008). In other words, the ability of an ecosystem to withstand stresses and shocks (and to continue to provide services) has a distinct asset value which can be degraded (or enhanced) over time.[2] Walker et al. (2010) look at the value of this resilience to agriculture in South-East Australia of maintaining a saline-free water table (mainly through farmers cutting down trees to expand agriculture). Here agricultural expansion represents a driver depleting the stock of non-salinated soils (measured as the depth of soils for which saline intrusion is not a problem). As this depletion driver is increased so the stock of ecological resilience falls. As the depleting process itself may generate benefits (here agricultural produce), there is a trade-off to be assessed between the benefits of depletion and the fact that losses of resilience may need to be reversed if stocks fall below some threshold level. Valuing this stock, unfortunately, is a relatively complex business, and extending this approach beyond largely illustrative examples is in its infancy at best. Indeed, Walker et al. are themselves extremely guarded about using their empirical example in the 'real world' owing largely to apparent uncertainties about the scientific and economic data. Nevertheless, such developments represent an important addition to existing ecosystem service valuation work. However, it is this existing body of evidence to which we now turn.

6.3 VALUING ECOSYSTEM SERVICES: LESSONS AND DIRECTIONS

The process of uncovering the true value of goods and using these data to ensure that decisions contribute to improving human well-being is a defining rationale for economic analysis.

[2] This approach can also accommodate a crucial concern about the nature of ecosystem assets: namely, that these resources are subject to threshold effects where services are subject to (possibly) greater risks of abrupt and extreme changes once a critical level of the asset has been breached.

A number of recent comprehensive reviews make clear the proliferation of methods—and applications of those methods—to assess the value of ecosystem services and biodiversity (see, for example, Pascual et al., 2010, US SAB, 2009, Bateman et al., 2011*b*, Kareiva et al., 2011). These assessments have been important for revealing, on the one hand, what is known about ecosystem and biodiversity valuation and, on the other hand, in identifying what we still need to learn. In what follows, we can only hope to provide a (partial) synopsis of these developments but, in doing so, we alight on a number of issues that strike us as noteworthy.

6.3.1 Economic valuation methods: a synopsis

There are many comprehensive reviews of economic valuation methods more generally (e.g. Bateman, 2002; Champ et al., 2003; Freeman, 2003; Pearce et al., 2006; Hanley and Barbier, 2009). Table 6.1 provides a brief overview of the key approaches. What is important to note here is that *all* of these methods have been used in the ecosystems context. In large part this breadth of methods reflects, in turn, the diversity of services that practitioners have sought to value rather than variety for its own sake.

The starting point for thinking about the valuation of ecosystem services is that such assessments rely on standard economic theory but with an underpinning by the natural sciences (Daily, 1997; MA, 2005; Pagiola et al., 2004; Heal et al., 2005; Barbier, 2007; Sukhdev, 2008). Whether this valuation can be based on market prices, or whether we must look to evidence from non-market behaviour (be this actual or intended) depends on the characteristics of the ecosystem good or service in question. In some cases, valuation might begin with market prices. For example, provisioning services are frequently market goods or near-market goods with close (market) substitutes. It follows, therefore, that market-based valuation has been prominent in such contexts, although perhaps these observed prices have needed to be adjusted for distortions (Table 6.1). However, the provisioning service is itself typically determined by some underlying service provided by an ecosystem process. Thus while the valuation of this final output is relatively straightforward, the analytical heavy-lifting is often done through the specification and estimation of an ecological production function. In other words, ecosystem services are frequently valued as a productive input (see Barbier, 2007; Freeman, 2003; Hanley and Barbier, 2009). In this approach, an attempt must be made to isolate and uncover the value of ecosystem services from the perspective of their effect on some observed level of output (Table 6.1). This approach can be applied to a range of market (consumption) goods but has also been used for valuing regulating and 'protection' goods (where examples of the latter include flooding and extreme weather protection).

Table 6.1. Summary of economic valuation methods used in ecosystem service valuation

Valuation method	Description	Typical applications to ecosystem services
Adjusted market prices	Using market prices adjusted for any distortions (e.g. taxes, subsidies, non-competitive practices)	Crops, livestock, woodland
Production function methods	Estimation of an ecological production function where the ecosystem service is modelled as an input to the production process and is valued through its effect on the output	Maintenance of beneficial species, maintenance of agricultural productivity, flood protection
Revealed-preference methods	Examining actual expenditures made on market goods related to ecosystem services. When market goods are substitutes, avertive behaviour or mitigating expenditure approaches can be used (e.g. expenditures to avoid damage, such as buying bottled water or installing double glazing). Travel-cost methods can be used when market goods are complements (e.g. travel costs for recreation). When the ecosystem service is a characteristic of the market good, hedonic price methods can be used (e.g. looking at the impact of noise or amount of green space on property prices)	Water quality, peace and quiet, recreation, amenity benefits
Stated-preference methods	Using surveys to elicit willingness to pay for an environmental change (contingent valuation), or to ask individuals to make choices between different levels of environmental goods at different prices to reveal their willingness to pay (choice modelling)	Water quality, species conservation, air quality, non-use values

In other cases, however, the value that people place on ecosystem services is not adequately reflected in market prices if at all. In such cases, non-market valuation techniques must be employed and applied to some ecological end-point which itself may have been estimated following some application of a production function. Revealed-preference methods value non-market environmental goods by examining the consumption of related market-priced private goods. A number of variants of the revealed-preference approach exist, depending on whether the environmental good and the related market good are complements, substitutes, or one is an attribute of the other (Table 6.1). In the first case, economists make use of the 'weak complementarity' concept introduced by Mäler (1974) to examine how much individuals are prepared to spend on a private good in order to enjoy the environmental good, thereby revealing the value of the latter. For example, the travel cost method examines the expenditure and time that individuals are prepared to give up to visit natural areas for recreation. In cases of substitutability between goods, approaches such as avertive behaviour or mitigating expenditures to avoid damages can be used, such as buying bottled water to avoid drinking contaminated water. Finally, the hedonic property-price method assumes that we can look at the housing market to infer the implicit value of the underlying characteristics of domestic properties, be these structural, locational/accessibility, neighbourhood, or environmental (Rosen, 1974). It can be used, for example, to examine the premium which people are prepared to pay in order to purchase houses in areas with greater proximity to green spaces or habitat types (Gibbons et al., 2011).

While revealed-preference methods estimate original values by looking at *actual* behaviour, eliciting values by looking at *intended* behaviour is the province of stated-preference (SP) methods. This is an umbrella term for a range of survey-based methods that use constructed or hypothetical markets to elicit preferences for specified changes in provision of environmental services (Table 6.1). By far the most widely applied SP technique is the contingent valuation method (see, for example, Alberini and Kahn, 2006).[3] However, in recent years, choice modelling has become increasingly popular. In this variant, respondents are required to choose their most preferred out of a (possibly relatively large) set of alternative policy or provision options offered at different prices, and their willingness to pay is revealed indirectly through their choices (see, for example, Hanley et al., 2001; Kanninen, 2007).[4]

In theory, SP approaches should be applicable to a wide range of ecosystem services and can be used to measure future/predicted changes in those goods. Importantly, such methods are thought to be the only option available for estimating those services which are valued for 'non-use' purposes. In practice, SP methods are mostly defensible in cases where respondents have clear prior preferences for the goods in question or can discover economically consistent preferences within the course of the survey exercise. Where this is not the case then elicited values may not provide a sound basis for decision analysis. Such problems are most likely to occur for goods with which individuals have little experience, or of which they have poor understanding (Bateman et al., 2008*a*,*b*, 2010). Therefore, while stated preferences may provide sound valuations for high-experience, use-value goods, the further we move towards considering indirect use and pure non-use values, the more likely we are to

[3] For a summary, see Carson's (2011) bibliography of published and unpublished contingent valuation studies from around the world.

[4] A number of studies combine revealed-preference (RP) and SP approaches in order to enhance the respective strengths of these data and minimize limitations (see, for example, Adamowicz et al., 1994).

encounter problems. Paradoxically then, where SP techniques are most useful is also where they have the potential to be less effective.

A number of solutions have been proposed for the problem of valuing low-experience goods. Christie et al. (2006) have proposed the use of intensive valuation workshops where participants learn about the environmental services being valued. However, the techniques involved are almost inevitably prone to reliance upon small unrepresentative samples which, after such intensive experiences, cannot be taken as reflecting general preferences. So while offering useful insights about overcoming the low-experience problem, it must be asked whether the cure is worse than the disease. Others have proposed and implemented extensions of conventional, individual-based SP applications. Bateman et al. (2009), for example, use virtual-reality software to convey images of landscape goods. This avoids the difficulties of conveying attributes of goods such as landscape in unfamiliar units such as hectares. Results show a significant reduction in the rate of preference inconsistencies through the application of such techniques.

Significant strides have been made in filling out the ecosystem valuation matrix without recourse to what might be judged by some to be more 'problematic methods'. However, an important finding of most assessments of this evidence base is that crucial gaps remain in the empirical record. One illustration of this is the case of cultural ecosystem services. Such services include use-related values such as leisure and recreation, aesthetic and inspirational benefits, spiritual and religious benefits, community benefits, education and ecological knowledge, and physical and mental health. Some of these services, such as recreation, are arguably straightforward to value. However, the challenge here is rolling out the available evidence where there is substantial spatial variation in the recreational value of ecosystems. Other categories of service are more difficult to measure at all—for example, those bound up by non-use motivations[5] such as altruistic, bequest, and existence values (Krutilla, 1967). Moreover, it is difficult to identify some of these benefits separately, or the specific contribution of ecosystems. In the case of the latter, this problem also characterizes efforts to establish the health–ecosystems linkage. To illustrate these challenges we consider further some of these issues.

[5] An existence value can be derived from the simple knowledge of the existence of the good or the service. In the context of the environment, individuals may place a value on the mere existence of species, natural environments, and other ecosystems. If an individual derives well-being from the knowledge that other people are benefiting from a particular environmental good or service, this can be termed altruistic value. Such values accrue during an individual's lifetime, but vicarious valuation can also occur intergenerationally. The effect on well-being of knowing that one's offspring, or other future generations, may enjoy an environmental good or service into the future, such as a biodiversity-rich forest being conserved, is termed bequest value.

6.3.2 Health values

Despite increased recognition that ecosystem services can have substantial effects on human health, both directly and indirectly (e.g. Myers and Patz, 2009; Bird, 2007; De Vries et al., 2003; Hartig et al., 2003; Mitchell and Popham, 2008; Osman, 2005; Takano et al., 2002; Ulrich, 1984), our knowledge of the complex relationships linking the biophysical attributes of ecosystems with the many aspects of human health remains limited (Daily et al., 2001).

Environmental quality and proximity to natural amenities is increasingly recognized as having substantial effects on physical and mental health, both directly and indirectly. Broadly, this could arise in a number of ways. Ecosystems provide many services that sustain human health (such as nutrition, regulation of vector-borne disease, or water purification). Also, natural settings could act as a catalyst for healthy behaviour, leading, for example, to increases in physical exercise, which affect both physical and mental health (Pretty et al., 2007; Barton and Pretty, 2010). Finally, simple exposure to the natural environment, such as having a view of a tree or grass from a window, can be beneficial, improving mental health status (Pretty et al., 2007) and physical health (Ulrich, 1984). Health outcomes in this respect can be disaggregated into two categories: reductions in mortality and reductions in morbidity (including physical and mental health).

While there is a large literature on health valuation, there is a crucial gap in relation to the contribution of ecosystems to these improvements. Moreover, the statistical evidence for the health–ecosystem link is still to be established unequivocally. For example, on the link between physical exercise and availability of green spaces, the suspicion is that even if the physical health link can be more firmly established, the value is possibly likely to be small given the availability of substitutes for this physical exercise. Hence, it is more likely to be the mental health benefit that is plausibly the more substantial of these two (bundled) health outcomes. Less is known as regards valuation here, although it might be the case that life satisfaction approaches linked to monetary valuation are a promising path to explore further (see, for example, MacKerron and Mourato, 2011). A final, but no less important, challenge is to know what values are for *changes in* ecosystem provision; most work to date has examined only the possible health benefits associated with *current* provision.

6.3.3 Non-use values

Environmental non-use values are often thought to be substantial (see, for example, Hanley et al., 1998). Critically, however, when and where these arise remains the subject of some discussion. Due to their intangible nature and disconnect from actual uses, the valuation of non-use benefits is complex. As a result there appears to be no systematic body of evidence about non-use values and, importantly, little consensus about how the empirical record (such as it is) can be used for practical assessment in the context of project (and policy) appraisals or broader national-level ecosystem assessments. In the former, a particular concern might relate to whether a (change in a) non-use value relates to a specific and discrete proposal (or the provision of a service more generally). In the latter, a concern might be double-counting or erroneously assuming that the same (per household or individual) non-use value estimate applies to all of the parts rather than something more broadly resembling the whole. Put another way, the physical 'unit' to which these non-use values apply is, on reflection, not at all obvious. Yet given the possible importance of non-use value in certain ecosystem contexts, this issue surely merits further investigation.

One significant obstacle to addressing this challenge is that, as already noted, SP methods are often thought to be the only economic valuation techniques capable of measuring non-use values, and so any doubts about the application of those methods or the accuracy of such valuations will loom especially large in this context. Challenges in the application of SP methods to non-use values are readily identified. Lack of experience and familiarity are likely to be important when respondents, for example, are asked about their preferences for non-use biodiversity species which might well be located in distant lands. Related to this is the lack of adequate testing for preference consistency exhibited in many such studies (although, for an exception, see Morse-Jones et al., 2012, discussed in further detail later in the chapter).

Other avenues for non-use valuation remain to be explored. For example, legacies can be argued to represent a pure non-use value. That is, individuals leaving a charitable bequest to an environmental organization in a will, for the purposes of supporting conservation activities, clearly will not experience the benefits of this work. Atkinson et al. (2009) estimate that, while (in 2007) only 6 per cent of all deaths in Britain resulted in a charitable bequest, their value remained substantial. And while legacies to environmental charities will be a relatively small proportion of this total, Mourato et al. (2010), for example, have estimated that this amounted to more than £200 million in the (financial) year 2008/09. Of course, legacies reflect only non-use values in the marketplace at the time of death. Moreover, data on charitable giving to recipient organizations, or according to demographic characteristics of donors, is not easily accessible, particularly for analysis over time. This is indicative of a wider problem. No approach appears to offer a general panacea for the challenges inherent in measuring non-use.

Related to the notion of 'non-use' is current interest in what has been termed 'shared values' (see, for example, Fish et al., 2011). For some this appears to be unfinished business arising from earlier discussions about how people value environmental policy changes, more generally, as individuals or citizens (Sagoff, 1988). However, the concept has also been a way of conveying that there might be something extra to the value of an ecosystem over and above adding up different elements of its total economic value.[6] The emphasis on shared values traces this missing element of value to the way in which ecosystems have collective meaning and significance for communities of people, related perhaps to 'non-use' or perceptions about ecosystem aesthetics.

There is less obvious evidence to add empirical substance to these insights. However, the handful of studies that have sought to use deliberative monetary valuation approaches provide some practical understanding of the individual or collective value of certain proposed environmental changes in a group context (e.g. Macmillan et al., 2002; Álvarez-Farizo et al., 2007), although our aforementioned comments about the representativeness of such findings still stand. Investigating this notion of shared values for ecosystems through wider-scale testing than has been possible thus far is a possibly rich topic for further research.

As an indication of the direction in which such reasoning might proceed, one reinterpretation of the 'shared values' argument is that it is a confusion between the individual making decisions on their own behalf and the same individual acting as social planner. In both cases the economic model applies directly but the beneficiary and hence the objective changes. Such a perspective is inherent in the contrast between the personal utility maximization problem faced by the individual (or profit maximization by a firm) and the optimization of net present value within a social cost–benefit analysis. A further source of confusion can arise from the observation that individual preferences are highly likely to be, at least in part, social constructs. Put another way, social context moulds individual values.[7] Under such an interpretation, the necessity of inventing new ways to measure apparently elusive 'social values' evaporates, to be replaced by a recognition that (i) the value of goods to an individual (who, for example, may bear only a fraction of any associated externality) may differ radically from the value of the same good from a societal perspective; and (ii) even those former individual values are highly likely to be in part the product of social (and other) contexts. None of this undermines the usefulness of social knowledge in the valuation process. Rather it provides a framework for the incorporation of such understanding within the decision system (uniting natural science, economics, and social

[6] Arrow et al. (2000) have made an analogous point in the context of the physical processes that the value of some system as a whole may be more than the value of the sum of its parts, perhaps because of complex ecological interactions.

[7] In much in the same way, that is, as a move across locations, and consequent environments, will alter the value of any given resource—e.g. water in the desert has a much higher marginal value than in areas of high rainfall.

science), and shows that such knowledge is vitally important if we are to understand the meaning and decision relevance of values and how they may alter between contexts; an issue to which we now turn.

6.3.4 Value transfer and spatial variability

Complex valuation processes, such as many of those involving ecosystem services, can themselves involve significant costs. It is therefore not surprising that a considerable literature has now evolved around the transferral of value estimates for environmental resources (Brouwer, 2000; Boyle et al., 2010) as a proxy for original primary valuation.[8] Although all value-transfer techniques involve the extrapolation of information from one context to another, Navrud and Ready (2007) identify two general approaches.[9] The simplest of these is to transfer mean values from some pre-assessed 'study' to the 'policy' context in question (see, for example, Muthke and Holm-Mueller, 2004). Such univariate transfers are frequently used in practical decision-making, but their validity depends crucially upon the significance of differences between the study and policy contexts, which should be small for transfer errors to be minimized. Clearly at some level all sites are dissimilar (e.g. the unique ecosystem habitats or the spatial pattern of substitutes around a site are unique). However, it is the degree to which this dissimilarity affects values which will determine the appropriateness of such 'univariate transfer' techniques.

The principal alternative to the univariate approach is to use statistical analyses to estimate value functions from study context data and to transfer those functions to policy contexts. This approach implicitly assumes that the variables determining the value of a good in one context will be the same as those affecting value in another context. Furthermore, it assumes that the relationships between variables and values will hold constant (i.e. in an estimated value-transfer function the list of explanatory variables and their coefficients are assumed to stay constant across the study and policy contexts). However, while parameters are kept constant, the values of the explanatory

[8] The bulk of this literature concerns the transfer of valuation estimates for improving some environmental resource. As such actions generate positive values so the literature is often labelled under the general heading of 'benefit transfers', a term which is frequently extended to the methods applied to effect such transfers. However, such terminology is somewhat confusing as these techniques are typically also valid for the estimation of costs associated with resource losses. A more accurate and general term is therefore 'value transfers', irrespective of whether a given application concerns the estimation of benefits or costs.

[9] The development of such approaches can be traced through Desvousges et al. (1992), Bergland et al. (1995), Brouwer and Spaninks (1999), Zandersen et al. (2007), and Johnston and Duke (2009). Other variants include meta-analysis (e.g. Bateman and Jones, 2003; Lindhjem and Navrud, 2008) and Bayesian approaches to modelling value functions (e.g. Moeltner et al., 2007; Leon-Gonzalez and Scarpa, 2008).

variables to which they apply are allowed to vary in line with the conditions characterizing each context. The value-transfer approach does not therefore look for similarity. Instead it looks for heterogeneity so as to capture the variety of factors which determine values. Differences between sites become prime drivers of consequent variations in estimated values.

One of the largest ecosystem services value-transfer exercises conducted to date forms the core of the economic analysis underpinning the UK NEA (UK NEA, 2011). Here value functions were estimated for multiple ecosystem services, including the provisioning value of agricultural food production, the regulating services of the environment as a store for greenhouse gases, and the so-called cultural services of both rural and urban recreation (including urban greenspace benefits). Following Bateman et al. (2011*c*), the functions were simplified to focus on the main—theoretically expected—drivers of value, thereby avoiding the transfer of factors which apply only in a given context and are not general. The functions were also built in an integrated manner which linked the levels of each to the other. So, for example, if provisioning values are increased as a result of agricultural intensification, that same intensification feeds into an increase in greenhouse gas emissions and deterioration of rural recreation resources, which result in a fall in both of these latter values. An example of the output obtained from such analyses, Figure 6.1 illustrates findings from the UK NEA analysis of rural recreation benefits arising from a change of land use from conventional farming towards multi-purpose, open-access, woodland (and discussed in the introduction to this chapter).[10] The distribution obtained by transferring a recreational value function across the entirety of Wales reflects various factors, including the distribution of population (this being highest in south-western Wales and in the areas of England neighbouring the north-east) and the availability and quality of the road network. Such spatially disaggregated outputs clearly allow decision-makers to target resources in the most efficient manner; an ability that is clearly of great importance during times of austerity.

Basing these integrated value-transfer exercises on highly disaggregated, spatially sensitive, large observation databases provides decision-makers with a rich and more holistic picture of the overall consequences of any given policy option. The advantages of such an approach were quickly realized by UK policy-makers, and the lessons of the UK NEA were explicitly incorporated in the UK Natural Environment White Paper (Defra, 2011), published in the immediate aftermath of the former report. Such academic and policy developments suggest that prospects for the incorporation of value-transfer techniques within institutional decision frameworks show promise. Notwithstanding this interim conclusion, there remains a need for tools capable of translating valuation information into policy action. We discuss this further in the next section.

[10] This in turn builds on Bateman et al. (2003).

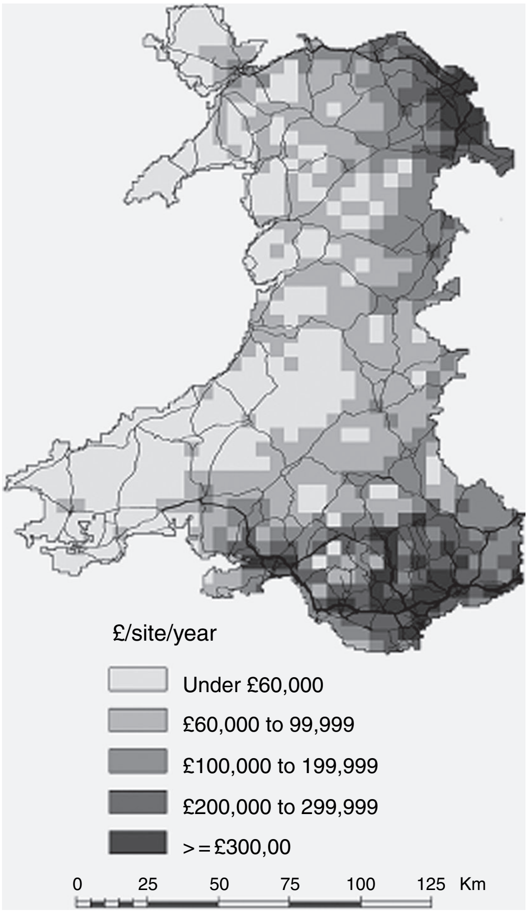

Fig. 6.1. Recreational values arising from a change in land use from farming to multi-purpose open-access woodland in Wales

Source: Adapted from UK NEA (2011).

6.4 FROM VALUES TO ECOSYSTEM ASSESSMENTS AND POLICY IMPLEMENTATION

6.4.1 Ecosystem valuation in the aggregate

The recent emphasis on large-scale ecosystem assessments—such as TEEB and the UK NEA—indicates some interest in searching for clues about the overall scale (in economic terms) of what has been lost (and what is likely to be lost in the future) as a result of the continued destruction of the natural world. While this is not a substitute for more detailed policy analysis, knowledge about these trends might be important for framing policy thinking. In addition, such information might throw light on whether ecosystem and biodiversity decline is a development problem as, for example, Stern (2007) demonstrated in the case of climate change.

One relatively long-standing insight here is that *particular* groups appear to be vulnerable to the loss of ecosystem services. Specifically, a number of studies have highlighted the dependence (of at least some portion) of the rural poor in the developing world on services provided by nature. Ten Brink (2011) terms this the 'GDP of the poor', although its antecedents can be traced to previous empirical studies of livelihoods, including Jodha (1986) and Vedeld et al. (2004). These studies have been important in highlighting the value of ecosystems to such communities (and well-being more generally), but their value is typically only partially reflected in official statistics. Less is known more generally, in either a developing or developed country context, about the way in which *aggregate* trends in, for example, ecosystem services and assets influence development (and development prospects). On the face of it, this is perhaps surprising and certainly contrasts markedly with the use of valuation in the climate change context, which has been, if anything, almost too exclusively concerned with global impacts. It seems worth asking why the hesitancy to aggregate has been so marked in the ecosystem context, and also whether this matters.

With regard to the 'why', inevitably it must be mentioned that Costanza et al. (1997) have cast a long shadow over the thinking of the economics community in respect of this (ecosystem value) aggregation issue. Specifically, Costanza et al. (1997) sought to provide estimates of the global value of ecosystem services from (in effect) the entire stock of all ecosystem assets. In doing so, that study famously calculated that the value of services or the 'output' provided by the natural world, in 1994, was in the region of $33 trillion (i.e. substantially in excess of gross world income at that time).[11] Not surprisingly, substantial debate was generated in the wake of this striking result. And perhaps most vocal among the critics were environmental economists (Pearce, 1998; Bockstael et al., 2000; Heal et al., 2005). On the face of it, economists might be thought natural bedfellows of efforts to boost the profile of valuation practice. Yet, this issue of valuing the well-being provided by the entirety of the global flow of ecosystem services struck at the heart of the basic premise of economic valuation. Put another way, valuing total services assumes that our baseline is (in essence) the loss of all ecosystems, and is a task that is unlikely to be adequately completed using methods that instead tell us something about the marginal value of a change in the stock of

[11] This point estimate is calculated to lie within a possible range of $16 trillion to $54 trillion (in 1994 US$).

ecosystem assets.[12] Although it does not explain entirely the current (apparent) reticence to aggregate, unease about 'repeating the Costanza et al. error' cannot be ruled out altogether as a contributory factor.

In reflecting critically, in this vein, on the Costanza et al. contribution, Bateman et al. (2011*a*) note the paradox of the positive impact that this paper has had, more generally, in raising awareness of the economic value of the natural world. It seems worth asking, therefore, what has been lost by not answering these aggregate questions. Two recent studies have sought to revisit these issues, but do so by calculating losses in natural assets likely to occur according to possible policy scenarios (and hence in principle ask a more defensible question than that about the totality of the current service flow). Hussain et al. (forthcoming 2013) estimate the losses arising from recent past and projected future loss of the world's aquatic ecosystems (specifically wetlands, mangroves, and coral reefs). The present value of this loss over the period 2000 to 2050 (using a discount rate of 4 per cent) is reckoned in excess of \$2 trillion (in 2007 US\$) (with two-thirds of this accounted for by wetlands). The annualized value of this total change is just under \$100 billion (that is, the value of the loss of these ecosystem assets each year is estimated to be of this magnitude) which, in 2007 for example, was just 0.2 per cent of global gross income. Chiabai et al. (2011) conclude not entirely dissimilarly for the case of the loss of global forests over the same time period.

Needless to say, such global estimates of ecosystem loss require some heroic assumptions and generalizations. Indeed, for some critics, a search for a global value is a flawed project because of this. However, given these findings, a tentative conclusion is that pragmatic demand (for more highly aggregated indicators) and concerns about validity both point away from an emphasis on the global perspective. Greater practical significance, however, is to be found at the regional or country level. In the case of forests, for Brazil, estimated losses in natural wealth are found by Chiabai et al. (2011) to be substantial (as a percentage of the country's gross national income or GNI). Hussain et al. (forthcoming 2013) find that for aquatic ecosystems, for the South Asia region and for Indonesia, however, these annual losses in natural wealth were respectively 1.7 per cent and 4.0 per cent of GNI (in 2007).

These are magnitudes worth knowing more about. This would necessitate close scrutiny of the robustness of such estimates. The basic problem of accounting for the value of ecosystems can be put simply. It entails identifying a price or (unit) value and a quantity of (some change in) the provision of, for example, ecosystem service (Boyd and Banzhaf, 2007). An immediate challenge, however, lies in identifying the likely limits on how the available empirical record on ecosystem 'prices' and 'quantities' can be pulled and

[12] Only if the value of a marginal unit is constant is it straightforward to go from valuing a single unit to valuing whatever number of units a given policy will create or destroy.

stretched over the assorted ecosystem areas needed to make robust aggregate generalizations. The issue of spatial variability here is central. This includes properly accounting for variation in the supply characteristics—the type and extent of functions—of ecosystems, as well as demand characteristics—of the human population that consumes services that these functions give rise to. All this requires relatively sophisticated mapping and is demanding in information terms. However, it might be that at this national level (or sub-national levels) these issues become a little more tractable (see, for example, Kareiva et al., 2011).

There are clear signs of growing interest in this question. An example of this is the linkages being made between (recent and ongoing) ecosystem assessments and efforts to understand the way in which changes in natural wealth influence the sustainability of development through greening of national accounts (see, for example, World Bank, 2010; Arrow et al., 2010). The ongoing World Bank-led consortium WAVES project (Global Partnership for Wealth Accounting for the Value of Ecosystem Services) represents a practical application of this work to a number of proposed countries.[13]

Of course, much of what we currently term 'ecosystem services' may already be reflected in our national accounts. This is a point made recently in World Bank (2010). Examples of this might include the natural pollination services that (in effect) are capitalized in the value of agricultural land or the recreational opportunities that are (implicitly and in part) provided by natural areas. On this view, ecosystems support market activity in a number of important (but indirect) ways, and the accounting challenge is to correctly re-attribute the service value to the (ecosystem) asset which gave rise to it (Nordhaus, 2006). As a starting point, an emphasis on identifying what is already (somewhere) in the accounts has merit. In particular, given the traditional opposition by the national accountants to non-market valuation in relation to the accounts (Hecht, 2005), this starting point has a strategic benefit.

6.4.2 Valuation and policy

While economics can contribute greatly to guiding the valuation of ecosystem services, it can also shape thinking about the implementation of policies aimed at delivering such values. Unfortunately, at present, many of the policies employed to deliver ecosystem services fail to heed either evidence regarding the way in which values can vary over different patches of ecosystems, or the lessons of basic economic theory regarding incentives that actors have to

[13] <http://wwwr.worldbank.org/programs/waves>.

reveal truthfully their valuation of services that they might provide. An example is provided by the UK Entry Level Stewardship (ELS) scheme (Natural England, 2010), which offers a flat-rate payment to all farmers irrespective of their location.[14] Such schemes fail to target payments to those areas which yield the highest values, and provide no incentive for farmers to provide anything other than the basic level of land management consistent with the scheme. Similar approaches characterize much of the increasingly substantial payments made under Pillar Two of the EU Common Agricultural Policy.

Thus economic valuation of itself is insufficient to improve the efficient delivery of ecosystem services. A simple example illustrates the problem and how economic intuition can help. Suppose that policy-makers seek to reduce diffuse water pollution from farms through a payment for ecosystem services (PES) scheme. A first requirement is to undertake a valuation exercise identifying those river catchments (and areas within those catchments) where reductions of pollution are likely to generate the largest net benefits. This might identify, for example, farms in locations above the inlet to water supply reservoirs as those most important to target. Now our focus must switch to the efficient implementation of such policies. One rather naive approach might be to simply ask farmers to state the levels of compensation they require to move towards modes of production which avoid diffuse pollution. Of course, farmers have an incentive to strategically overstate their compensation requirements. However, the economic theory of auctions suggests that even relatively simple approaches can significantly improve implementation efficiency (Vickrey, 1961; Clarke, 1971; Groves, 1973; Groves and Ledyard, 1977). For example, switching to a simple sealed-bid contracting system might reduce the potential for strategic responses and improve incentive compatibility. This could be the case if farmers are told that contracts will be awarded according to the combination of pollution reduction and cost.

In certain circumstances even greater efficiency gains can be obtained. For example, where the delivery of ecosystem services can be readily measured (e.g. in policies seeking the provision of certain habitats) then landowners will be those best able to judge whether their land is particularly suitable for providing such goods (or faces the lowest opportunity costs). Such actors can outbid competitors by offering better outputs (or lower costs) than their rivals.[15] To date, practical examples of such agreements are, at least in the UK,

[14] An exception here is the minority of farms located above the 'Moorland Line' (Natural England, 2010), where a lower, but again flat, rate payment is available.

[15] Such markets can also be designed to benefit private sector purchasers of ecosystem services—for example, water companies may be able to reduce their costs of providing potable water by avoiding costly treatment options by engaging with landowners to reduce pollution inputs to rivers. Indeed economic theory identifies the potential for multiple private sector bodies to combine to purchase such services, provided that markets are created so as to avoid free

generally confined to the experimental laboratory. However, proposals have been made by a number of policy-makers (and indeed the authors) that the development of such implementation tools should be a major focus of the next phase of work under the UK NEA. The earlier example indicated that valuation, while typically necessary for good decision-making, is not in itself sufficient.

One further point is that valuing ecosystems and biodiversity is a complex endeavour and often at the frontier of valuation knowledge. This suggests good reason, in certain contexts, to be circumspect about the role that valuation might play in informing decisions about conservation. Decision-making in such situations where values are unknown—or where values cannot be established to any degree of validity—has generated much debate. In such cases, however, 'caution' (given what might be lost) might be a sensible watchword. Possible responses include the adoption of ecological standards sometimes termed 'safe minimum standards' to ensure the sustainability of resources which are not amenable to valuation (Farmer and Randall, 1998), or compensating offsetting projects validated for their ecological suitability (Federal Register, 1995). In such cases, the role for valuation might be a greater emphasis on cost-effectiveness in meeting specified targets.

An illustration of this challenge in determining how exactly valuation should guide social decision-making is provided by the example of valuing biodiversity. Weitzman (1993)—using the example of the world's remaining species of cranes—defines biological importance of each species in terms of their taxonomic distinctiveness (e.g. of the whooping crane compared with other crane species),[16] and the likelihood of extinction (of a given species). Assuming that maximizing (expected) diversity is our objective, species conservation becomes a problem of cost-effectively distributing the marginal (available) unit of money from conservation funds to where it achieves the highest pay-off. Typically, this will be where there is some combination of high diversity and low survival probabilities.

Ideally, it would be useful to extend such insights with reference to the preferences that people might have for diversity. Somewhat reassuringly, Morse-Jones et al. (2012), for example, find that SP responses reveal expected substitution patterns across ecologically similar species—e.g. different small amphibians. However, preferences need not always conform to what is ecologically feasible or sustainable. Thus, in the Morse-Jones et al. study, respondents had a massively stronger preference for iconic, physically large, and especially furry animals which dwarfs concerns regarding ecologically crucial

riding by ensuring that PES trades go ahead only if all parties contribute to their purchase (Güth et al., 2007; Potters, 2007; Eckel and Grossman, 2008; Bracht and Feltovich, 2008).

[16] Genetic distinctiveness is defined by Weitzman (1993) as the evolutionary distance each existing species is from a common ancestor species.

issues such as extinction threat. So, for example, willingness to pay to conserve lions, even where these animals are not threatened by extinction, hugely outweighs stated values for, say, a species of frog, even when it is on the brink of extinction. Another example is provided by Bateman et al. (2009). That study observes that while respondents had strongly positive preferences for enlarging an area of freshwater marshland suitable for visiting and viewing bird populations, they had negative values for an adjoining area of tidal mudflats, even though these were a major source of food attracting those birds to the area. In many respects, these findings are not surprising. However, what it does raise is a deeper question about the extent to which economic values can be a guide for decision-making, or whether ecological constraints need to be considered. Clearly, the claim that human preferences are (almost always) 'right' or 'wrong' is overly simplistic at either extreme. However, where to draw this line is far from obvious and—given changing knowledge—is anyhow likely to be a shifting target. Nevertheless, while recognizing the importance of economic values for thinking about the importance of ecosystems and guiding policy thinking, we need to be mindful of the complexities and uncertainties involved.

6.5 CONCLUSIONS

The valuation of ecosystem services has become a crucial element (perhaps *the* crucial element) in quantifying the contribution of ecosystems and biodiversity to human well-being. A significant body of research has already begun to emerge, and a number of recent national and international ecosystem assessments have helped provide further impetus to such efforts. Needless to say, significant challenges remain. Hence, while the evidence base is broad and on occasion deep, reflections on this literature in a variety of existing reviews have identified, for example: a need for greater understanding of ecological production, especially as it relates to spatial variability and complexities in the way that services are produced; the size and significance of inevitable gaps in the empirical record as well as the ability to fill these gaps by judiciously transferring values; and the scope and limits in using this evidence base to inform practical decision-making, both generally and in relation to concerns about whether the valuations that we find in this literature genuinely tell us about the importance of ecosystem assets and biodiversity.

In this chapter, we have sought to highlight some of these issues, although unavoidably our discussion cannot be exhaustive. Much of our focus has been on valuation methods and particularly the challenges inherent in seeking to value non-market costs and benefits. Some of these challenges involve general considerations, although other issues are specific to valuing ecosystems and biodiversity, or at least seem particularly acute in that context.

Such challenges need to be viewed in context. The recent UK NEA (NEA, 2011) has shown how the empirical record can be put to use in an informative and policy-relevant way. Thus, there are encouraging signs that value-transfer methods (i.e. transferring the empirical record to new policy contexts and questions) can be used in an increasingly effective manner. If so, concerns about whether we can adequately measure the way in which ecosystem values vary across space (because of geographical variability in the way that services are supplied by nature and valued by people) might be addressed. Such developments could be crucial in translating valuations into meaningful policy analysis. It may also offer some hope for shedding light on the value of what is lost when and if ecosystems and biodiversity are degraded and destroyed in more highly aggregated assessments. This is not just an issue of only identifying aggregate trends (for which policy uses would be limited, apart from perhaps raising the profile of conservation issues generally). There are fruitful linkages to be made about the way in which what is happening to (natural) wealth influences development paths.

Thinking about ecosystems as assets also helps identify some critically important issues that are arguably neglected in most of the valuation literature as it has been applied to ecosystem services. This relates to the way in which future services are valued when an ecosystem asset undergoes some change. While such questions are commonplace elsewhere, in the ecosystem context these have only begun to be asked, although related issues of valuing ecosystem complexity have a longer standing. Progress on these matters, both in theory and practice, is surely only a matter of time. Nevertheless, it seems unavoidable that uncertainties will remain. That is, while we can conclude positively on the rapidly evolving scope for ecosystem and biodiversity valuation to contribute to a profound understanding of suitable policy responses, there remains room for debate about whether valuation is in itself enough to ensure effective policies, as well as how to conduct decision analyses in those contexts where valuation and understanding of the natural world are likely to remain relatively uncertain.

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7

The Economics of Ecosystems and Biodiversity (TEEB): Challenges and Responses

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7.1 THE ECONOMIC CASE FOR BIODIVERSITY

If economic arguments could make such a strong case for early action and policy change to address the threat of climate change, then could the same be possible for biodiversity loss? This was in essence the question put forth by a group of G8 + 5 environment ministers in Potsdam, Germany, in 2007, referring to the recently published 'Stern Review of the Economics of Climate Change' (Stern et al., 2006). To explore this question further, an initiative known as 'The Economics of Ecosystems and Biodiversity' (TEEB <www.teebweb.org>) was launched by Germany and the European Commission. Half a decade after its genesis, this chapter describes the life of TEEB to date, progress made towards its goal of mainstreaming the economics of nature, the main challenges facing TEEB as it begins a phase of implementation, and the responses of the 'TEEB community' to these challenges.

The causes of ecosystem degradation and biodiversity loss were well documented in the Millennium Ecosystem Assessment (MA, 2005), which also listed the many kinds of values delivered to society and the economy by nature. The TEEB reports, which followed the MA's ecosystem service classification, compiled the available evidence and highlighted how these values often go unrecognized by decision-makers across society, be they policy-makers, administrators, businesses, or citizens. Because nature is often invisible in the economic choices we make, we have steadily been drawing down our natural capital—without understanding either what it really costs to replace services provided for free by nature, or that man-made alternative solutions are sometimes far too expensive for these services to be replaced or substituted. Exacerbating the problems associated with economic invisibility of nature and its services in most policy discourse and in policy trade-offs is the inadequacy of today's economic compass—comprising GDP and related indicators at the macro level, and financial profitability or 'shareholder value' at the micro level. These indicators are old, incomplete, and no longer capable of providing good answers in modern society, in a world where natural resource scarcity affects a diverse range of public and private goods and services.

TEEB is an initiative to compile the evidence on these problems in their biophysical and spatial contexts and their socioeconomic contexts, and also to address metrics for their evaluation and redressal. The purpose of the 'Interim Report' of TEEB (2008) was to size in economic terms the problem of ecosystem degradation and biodiversity loss. It was presented at the High-Level Segment of the ninth Conference of the Parties to the Convention on Biological Diversity (CBD COP-9) in Bonn, Germany, in May 2008, and sparked international demand for a deeper analysis of the economics of ecosystems and biodiversity. Responding to this call, the TEEB initiative embarked on delivering a series of reports focused on different groups of decision-makers. At CBD COP-10 in Nagoya, Japan, in October 2010, the last of five reports was presented: the first publication, 'TEEB Ecological and Economic Foundations', provided a comprehensive assessment of the fundamental ecological and economic principles of measuring and valuing ecosystem services and biodiversity. Aimed at policy-makers, the second report, 'TEEB in National and International Policy Making', and the third, 'TEEB for Local and Regional Policy Makers', offered targeted guidance on how investment in natural capital could deliver a wide range of social and economic benefits, and practical insight into which policy options exist to better manage these changes. The fourth report in the series, 'TEEB in Business and Enterprise', described how biodiversity loss and ecosystem decline present both risks and opportunities to businesses, and examined how businesses can align their actions with conservation goals by better recognizing and responding to their dependencies and impacts on ecosystem services. The final report provided a synthesis of the approach, conclusions, and recommendations of the initiative.

The TEEB suite of reports has quickly gained credibility as a leading, up-to-date source of knowledge in the discipline of ecosystem and biodiversity valuation. Despite its wealth of data on tools and methodologies, a conscious decision was made not to produce any aggregate number for quantifying either a single global value for nature's services or the global economic damage due to lost biodiversity,[1] as will be explained later in this chapter. Several

[1] Although the TEEB studies refrain from producing an aggregate number, they do occasionally cite and base their findings on other pieces of work that have made such attempts—for

factors influenced this choice, such as establishing the meaning or relevance of any such value given that we have no alternatives to Earth's biosphere; the plurality of ethical perspectives for valuation, its purposes, and its contexts; and, conversely, the actionability and human relevance of working at scales such as biomes, countries, regions, and communities.

Instead, with 'mainstreaming' as its avowed principal objective, TEEB intends to help decision-makers recognize the wide range of benefits of ecosystems and biodiversity, demonstrate their values in economic terms and, where appropriate, suggest how to capture those values in decision-making.

7.2 TEEB AND ECONOMIC VALUATION

Whilst inspired by the Stern Review, it was evident from TEEB's inception that the nature of the challenge being addressed by TEEB was different from climate change. Biological diversity, or biodiversity, refers to the entire living fabric of our planet—comprising its ecosystems, species, and genes,[2] in all their quantity and quality dimensions. This formalistic definition from the CBD, together with the work of the Millennium Ecosystem Assessment, helps us recognize the many levels at which nature's living fabric nourishes and sustains human societies and economies. Any study of the costs of 'business as usual', or any attempt to value the benefits of nature's services, needs to work across these different layers of biodiversity; across different geo-political scales at which benefits flow (local, regional, global); across different value-articulating institutions (TEEB, 2010*a*) and their valuation perspectives; and across different institutional spaces in which responses to loss and degradation can be formulated by society, ranging from norms, regulations, policies, and economic mechanisms, to markets.

All of these very different biodiversity layers, geo-political scales, value-articulating institutions, and diverse response strategies developed by decision-makers to address biodiversity losses *together* constitute the landscape of TEEB. Precisely because of the variances and vagaries of this landscape, TEEB cannot and thus does not propose a one-size-fits-all, cost–benefit-based stewardship model for the whole Earth. Instead, TEEB sees valuation as an important human institution (TEEB, 2010*a*). Douglass North defined 'institutions' as the basic rules of the game in an economy (North, 1990). These could either be formal systems, such as constitutions, laws, taxation,

example, Braat and ten Brink (2008), which contains an economic assessment of the value of biodiversity loss in 2050 compared with 2000, according to a business-as-usual scenario. Although arriving at monetary results, it cites numerous caveats, making the results partial and tentative.

[2] <www.cbd.int>.

insurance, and market regulations, or informal norms of behaviour, such as habits, customs, and ideologies. In the same way, the institution of valuation can also be informal or formal, depending on its socio-cultural context. In other words, valuation is a 'constructed set of rules or typifications' (Vatn, 2000), emerging from our understanding of what they are and how they should be determined. Values, norms, beliefs, and conventions are part of culture, and they can show considerable diversity, which in turn affects valuations (TEEB, 2010*a*, p.161). For example, Judaeo-Christian culture and beliefs see man as 'inheritor of Earth', as owner. However, such a view contrasts sharply with naturist or tribal views of humanity as part of the fabric of nature. TEEB argues that neither is incorrect nor invalid in their respective socio-cultural contexts, as values are always derived from worldviews and perceptions.

A basic premise of the TEEB (2010*b*) study is that the valuation of biodiversity and ecosystem services may be carried out in more or less explicit ways according to the situation at hand. The TEEB study follows a tiered approach in analysing and structuring valuation that involves three different levels of action (see next).[3] Although not all are necessary for ensuring conservation and sustainable use, and indeed some require more attention than others depending on context, a holistic approach is strongly encouraged:

1. **Recognizing value**: identifying the wide range of benefits in ecosystems, landscapes, species, and other aspects of biodiversity, such as provisioning, regulating, habitat/supporting, and cultural services;
2. **Demonstrating value**: using economic tools and methods to make nature's services economically visible in order to support decision-makers wishing to assess the full costs and benefits of land-use change; and
3. **Capturing value**: incorporating ecosystem and biodiversity benefits into decision-making through incentives and price signals.

All of these levels of valuation help us to rethink our relationship with the natural environment and alert us to the impact of our choices and behaviour on distant places and people.

'Recognizing value' is a capability of all human societies and communities, and can easily influence societal norms and regulations, often without any recourse to monetization or even economics. One such example is the tribal communities of Himanchal Pradesh, India, who protect thousands of sacred groves due to strong spiritual beliefs. Other examples come in the form of legislation, such as declaration of protected areas for reasons of patrimony and

[3] For a collection of nearly 100 case studies illustrating the TEEB approach, see the European Environmental Agency's 'Eye on Earth' website at <http://www.eea.europa.eu/atlas/teeb>.

heritage, thereby bequeathing unique areas for future generations to enjoy. Changes in land management and planning strategies in recognition of ecologically important areas are also examples of value recognition.

'Demonstrating value' in economic terms is critical for understanding the consequences of changes resulting from alternative land-use or land-management options, and can be an important aid in achieving more efficient use of natural resources. For example, an assessment in Kampala, Uganda compared the costs and benefits of conserving the ecosystem services provided by wetlands in treating human wastes and controlling floods against the costs and benefits of providing the same services by building water treatment facilities or concrete flood defences, and found the former to be considerably less expensive (Emerton et al., 1998). Demonstrating value can also highlight the costs of achieving environmental targets and help identify more efficient means of delivering ecosystem services. Valuation in these circumstances enables policy-makers to address trade-offs in a rational manner, correcting the bias typical of much decision-making today, which tends to favour private wealth and physical capital above public wealth and natural capital.

'Capturing value' can be achieved through a variety of economic mechanisms, some of which can be market-based (e.g. eco-labelling, eco-certification, and 'payments for ecosystem services' (PES)), whereas others are embedded in policy decisions. Legislation or liability rules can also work to incorporate values into the private and public sphere of decision-making. It is observed that, in the majority of PES schemes, both payers and receivers are government entities,[4] and this further highlights that value capture takes place in a much wider solution space, and is not the same as 'marketization' of the natural commons.

'Market' solutions assume commodification, many buyers and sellers, and the existence of *private* claims to buy and sell. However, most ecosystem services that are being degraded and most biodiversity that is being lost is categorized as *public* goods and services, for which markets are far from ideal vehicles of management.

7.3 RESPONDING TO THE CHALLENGES

There are four widespread and legitimate concerns about economic valuation of nature's services, each of which has been addressed by TEEB in the design of its own approach to undertaking valuation.

[4] See 'Scaling Up Biodiversity Finance: Co-chairs' Summary' (2012), Dialogue seminar, Quito, Ecuador (available at <http://www.dialogueseminars.net/resources/Quito/Report/Quito-report-8-April.pdf>).

First, valuation of nature necessarily involves a certain degree of subjectivity (Prior, 1998; Lockwood, 1999; Balmford et al., 2011). Values, as well as norms, beliefs, and conventions, are derived from worldviews and perceptions of a society that try to understand and delineate what is right or wrong or, more appropriately, what is invaluable, valuable, or valueless (TEEB, 2010*a*, p. 161). Because of this multi-dimensional and socio-cultural embeddedness of 'value', any exercise of valuation is purely a reflection of how certain people perceive their natural environment, and their relationship to it, at a certain point in time (TEEB, 2010*a*, p.151). This subjectivity is indeed recognized, and forms an important part of TEEB's approach to decision-making. While economic valuation can be a powerful means for decision-making and feedback, it is only one particular tool based on a rational management approach (TEEB, 2010*a*, p. 157). In situations where cultural consensus on values is strong, and the science is clear, valuation can contribute to more holistic economic accounting and planning, with an inclusive view of nature and its benefits. However, in complex situations involving multiple ecosystems and services, and/or plurality of ethical or cultural convictions, valuation data may be unreliable or unsuitable. In such cases a differentiated discussion of what choices society has regarding our relationship with nature and what risks these involve is all the more important. In general, TEEB advocates providing the best available estimates of value for a given context and purpose, and seeking ways to internalize that value in decision-making.

The second concern is derived from the view that values are generally incommensurable, in that they cannot be measured in the same units (Faucheux and O'Connor, 1998; Funtowicz and Ravetz, 1994; Martinez-Alier et al., 1998; Martinez-Alier and O'Connor, 1999; Sagoff, 1998). The very idea of valuation, however, exists on the dangerous premise that nature can be reduced to a single (usually monetary) metric, and is thus commensurable. This is akin to equating something like a human rights infraction or loss of life with financial compensation, and fails to take into account that certain values simply cannot be measured, such as intrinsic or existence values of nature (Gatzweiler, 2008, cited in TEEB 2010*a*, p. 162; Sagoff, 2011). This is indeed a serious concern, and any estimate of total economic value runs the risk of leaving out important aspects. It is therefore essential to communicate monetary values with diligence, making clear which dimensions they do and do not cover, and communicating them as lower boundary, not as 'true value'. TEEB itself goes beyond valuation and attempts to place nature's values in their appropriate context. TEEB acknowledges that economic trade-offs form an important part of policy-making, and that monetary valuation may be helpful in providing economic incentives to sustainably manage ecosystems (Costanza, 2006), or at the very least, trigger the much needed societal debate about the value of nature and its services beyond the conservation of birds and butterflies, considered by many as a luxury of the rich.

Third, there is a strong fear of adding economic uncertainty to ecological uncertainty, as TEEB presumes to operate in a space of scientific uncertainty about ecosystem services, and exacerbates risks by adding a layer of economic analysis to this uncertainty (Chee, 2004; Johnson et al., 2012). There is no doubt that there is a high level of uncertainty about the supply of natural resources and ecosystem services, especially into the future, and this makes economic valuation difficult if not contentious. Moreover, there is still a large (albeit narrowing) knowledge gap regarding the consequences of ecological and anthropogenic processes for the health and functioning of biodiversity and ecosystem services. Risks and uncertainty are innate to our modern world of complex and interrelated problems.

For instance, one of the biggest uncertainties facing economic analyses of biodiversity and ecosystems is the characterization of the responsibility of the present generation for the well-being of future generations. Selecting an appropriate discounting rate[5] is the outcome of explicit or implicit ethical choices and, much like the Stern Review's economic analysis of climate change, the loss of biodiversity and ecosystems has properties that make it difficult to apply standard welfare analysis, including discounting the future:

(i) It is a phenomenon having global, regional, as well as local consequences.
(ii) Its impacts are long-term and irreversible.
(iii) Pure uncertainty is pervasive.
(iv) Changes can be non-marginal and non-linear.
(v) Questions of both inter- and intra-generational equity are central.

TEEB approaches this dilemma by presenting a range of discounting choices linked to different ethical standpoints, thereby enabling end-users to make their own conscious choices. The use of positive rates is supported by the view that goods or services delivered later are relatively less valuable when incomes are expected to grow, even though this will typically lead to the long-term degradation of ecosystems and biodiversity; a discount rate of zero translates into a more ethical approach that typically sees our grandchildren valuing nature similarly to our generation, and deserving as much as we do; even the use of negative rates can be applied under the assumption that future generations will be poorer in environmental terms than those living today. Generally speaking, TEEB advocates that a variety of discount rates be considered depending on the time period involved, the degree of uncertainty, ethical responsibilities to the world's poorest as well as future generations, and the scope and nature of the project or policy being evaluated.

[5] For a detailed discussion of discounting the future in an ecosystems and biodiversity context, see TEEB (2010*a*), 'Chapter 6: Discounting, Ethics and Options for Maintaining Biodiversity and Ecosystem Integrity'.

However, it must be mentioned that, in situations characterized by non-marginal change, radical uncertainty, or ignorance about potential tipping points, economic valuation tends to be less useful. In such circumstances, prudent policy should invoke complementary approaches such as the 'safe minimum standard' or the 'precautionary principle'. TEEB argues that the most ethical response for us in the face of risk and uncertainty is not to sit idly until we have perfect information to act. As a society, we are confronted with a moral choice of whether or not to act. TEEB considers the economic perspective as complementary to all others and, after compiling all of the evidence, sees risks and uncertainty in the context of the equally if not more serious risks and uncertainties of proceeding along a 'business as usual' path, despite all available evidence that nature's losses are palpable, serious, harmful, and potentially disastrous for human survival in the biosphere. Given the choice between the increasing present and future costs of inaction or the long-term benefits of imperfectly informed action, the preference of the TEEB community is to err on the side of caution and conservation.

Lastly, there exists a concern that we are 'selling the rights of Mother Earth'[6]—in other words, that the 'financialization' (Spash and Aslaksen, 2012; Arsel and Büscher, 2012; Sullivan, 2013) of nature and its services will ultimately lead to its commodification and marketization (Khor, 2011; McAfee, 1999; McCauley, 2006). More specifically, this criticism suggests that nature, once its values are identified and expressed in monetary terms, will become a market commodity and, like any other, subject to free trade. Moreover, it is argued that, in becoming privatized, previously public ecosystem goods and services will become accessible to the very same private interests responsible for our planet's degradation (Monbiot, 2012). Though these are valid concerns, we would, however, argue that essential ecosystem services are already being 'traded' in precisely this manner, sometimes for an implicit price of zero (Costanza et al., 2012). Land concessions granted for mining or logging usually do not account for the ecosystem services lost through subsequent land-use change. Ocean commons continue to be open-access and free. If nothing else, valuation in combination with liability regulations makes destructive extraction less attractive by adding (usually quite significant) financial costs. Placing a value on nature's ecosystem services should not be misconstrued as 'putting a price on nature'. Economic valuation utilizes several instruments—some market-based and some not—to reflect the

[6] This fear is most typically voiced by members of ALBA countries. ALBA, or The Bolivarian Alliance for the Peoples of Our America (Spanish: Alianza Bolivariana para los Pueblos de Nuestra América), is an international cooperation organization for the social, political, and economic integration of the countries of Latin America and the Caribbean. Member nations include Antigua and Barbuda, Bolivia, Cuba, Dominica, Ecuador, Nicaragua, Saint Vincent and the Grenadines, and Venezuela. These views are reflected in an Open Letter to the CBD, available at <http://www.wrm.org.uy/countries/Ecuador/Open_Letter_Global_Dialogue_Seminar.html>.

value of nature's services.[7] TEEB does not suggest placing blind faith in the ability of markets to optimize social welfare by privatizing the ecological commons and letting markets discover prices for them. What TEEB offers is both a model for communicating to decision-makers in their own language, dominated by economics, as well as a toolkit for evaluating and integrating good stewardship into their decisions.

A whole range of policy and legislative responses is required to solve the largely public goods problems underlying biodiversity loss and ecosystem service degradation across different countries and societies—such as changes in land-use planning, regulation changes, community access rights reforms, eco-labelling and eco-certification, valuations of protected areas' benefits, schemes for payments for ecosystem services, to name a few. Most importantly, as a society we have to reopen the debate on our relationship with nature, the choices that we are facing and the options that we have. The fundamental problem of biodiversity loss can be addressed only if we find new ways of explicitly debating about value and importance. In such a debate, valuations (understood in the broad sense explained by TEEB reports, rather than a narrow sense of 'marketization') can be very useful in providing substance and credibility to arguments for better conservation policy and practice. But the debate should by no means be limited to our current understanding of valuation, and should also explicitly address drawbacks and limitations as this will help achieve a much more encompassing debate, where economics is a means to the end of achieving human well-being.

7.4 TAKING TEEB FROM ANALYSIS TO ACTION

Capitalizing on the step-change in awareness created by the TEEB reports, TEEB has become increasingly recognized and explored as an essential toolkit for decision-makers in governments and business to integrate the economic value of biodiversity and ecosystem services into their accounting and reporting systems. Its ongoing phase of implementation is taking TEEB into a growing number of countries and into a very broad-based 'TEEB for Business Coalition', comprising several global business networks. Progress thus far is very much in line with TEEB's central objective of 'mainstreaming' the economics of ecosystems and biodiversity; however, these are early days and significant challenges lie ahead, not least the need to ensure that sufficient checks and balances and careful planning address inappropriate use of valuations.

[7] For example, subsidies, regulation, investment in public goods/ecological infrastructure, distributional impacts, and poverty eradication incentives.

The role of the TEEB initiative in this third phase is to support policy-makers and the world of business in their efforts to undertake TEEB studies, and to better respond to ecosystem degradation and biodiversity loss through policy instruments and reforms. A TEEB study can be undertaken at the regional, national, or sub-national level, in both public- and private-sector contexts. It can cover different issues and ecosystems, incorporate different types of information, and should consider a wide range of stakeholder perspectives. Therefore, there is (and should be) no single valuation process that can be applied to every situation. Instead, TEEB has analysed many cases and, from this analysis and the broader literature, summarized a stepwise approach consisting of six steps (see Box 7.1) to help structure the process of explicitly assessing and incorporating ecosystem services into policy and management decisions. These steps should be integrated into and inform the usual processes in decision making and policy design established in different countries and are intended to complement not to replace these.

These steps are integral to the operationalization of TEEB and have quickly been picked up by regional and national authorities in order to establish their own TEEB studies.[8] National and local governments have an essential role to play in this process, whether by mainstreaming biodiversity and ecosystem services into policy-making, or by creating an enabling regulatory and fiscal environment for business. Appreciating the responsibility that this entails for ensuring quality, TEEB's Advisory Board recently set up a process whereby country-level TEEB studies can undergo a structured peer-review process and, once reviewed by a Board committee of experts, can then be endorsed as a recognized 'TEEB Country Study'.[9] Moreover, in the international policy-making setting, TEEB is featured prominently within intergovernmental strategies and processes on biodiversity and ecosystem service issues.[10]

The private sector plays a crucial role in influencing biodiversity loss, although its responses are not generally commensurate with its impacts. Although many companies now report their greenhouse gas emissions and mitigation efforts, biodiversity and ecosystem services are usually treated superficially in company reports, and are rarely seen as relevant to financial reporting. However, the business case for biodiversity and ecosystem services is getting stronger as resources become scarce, and market opportunities shift towards green

[8] TEEB studies and assessments are currently under way in several regional (e.g. Association of South-East Nations, or ASEAN, European Union, and Nordic countries) and country-level contexts (e.g. Brazil, Georgia, Germany, India, Netherlands, Norway, South Africa, St Lucia, and Sweden), as well as in the context of European Commission pilot projects in Bhutan, Ecuador, Liberia, the Philippines, and Tanzania.

[9] A 'Guidance Manual for TEEB Country Studies' was launched in May 2013 and provides both technical and operational guidance on how countries may conduct a TEEB Country Study. It outlines the various steps that may be taken to initiate and implement a country study, communicate its findings, and implement the recommendations of the study. It can accessed at <http://www.teebweb.org/wp-content/uploads/2013/06/TEEB_GuidanceManual_2013_1.0.pdf>.

[10] Examples include the CBD Strategic Plan for Biodiversity 2011–2020 and its Aichi Biodiversity Targets (particularly 2, 3, and 11), EU Biodiversity Strategy to 2020, and the IUCN Programme for 2013–2016.

Box 7.1. **The TEEB stepwise approach**

Step 1: Specify and agree on the problem
This is often a worthwhile effort because views can differ substantially. If key stakeholders share a common understanding of the problem, serious misunderstandings during the decision-making process and implementation can be avoided.

Step 2: Identify which ecosystem services are relevant
Ecosystem services are often interconnected. Identifying which ones are most important to your problem focuses the analysis. Going one by one through the list of services is a simple approach.

Step 3: Define the information needs and select appropriate methods
The better you can define your information needs beforehand, the easier it is to select the right analytical method and interpret the findings. Assessments differ in terms of which services are considered, the depth of detail required, timelines, spatial scope, monetization of the results, and other factors. The study design determines what kind of information you get.

Step 4: Assess expected changes in availability and distribution of ecosystem services
If possible, use experts. Also, draw on field work and documented experience from analyses in comparable settings. Use common sense and consult with colleagues on possible changes and their consequences, starting with the most obvious ecosystem services.

Step 5: Identify and appraise policy options
Based on the analysis of expected changes in ecosystem services, identify potential responses. Appraise these in terms of their legal and political feasibility as well as their potential in reaching the targeted quality, quantity, and combination of ecosystem services produced by natural capital.

Step 6: Assess distributional impacts of policy options
Changes in availability or distribution of ecosystem services affects people differently. This should be considered in social impact assessment, either as part of the analysis or as part of appraising policy options.

The relative importance of each step is determined by your situation and objectives. Taken together, adapted to specific needs, and incorporated into existed decision-making procedures, they offer guidance for considering natural capital in local policy. Other technical, legal, economic, and social information also needs to be considered. The steps can also help design a monitoring system and thereby track the condition of natural capital.

Source: TEEB (2010*a*), pp. 38–41, adapted from WRI (2008).

businesses. Companies that understand and manage risks presented by biodiversity loss and ecosystem decline, establish operational models that are flexible and resilient to these pressures, and move quickly to seize business opportunities, are considered more likely to thrive in future scenarios. TEEB offers a number of reliable tools and methods for determining the economic value of nature's services, which can in turn be used, for and by business, to help make the link from ecological impacts and dependence to the business bottom line.

Corporate externalities—i.e. unaccounted costs to society of doing 'business as usual'—of just the top 3,000 listed companies amount to an estimated US$2.15 trillion, or 3.5 per cent of GDP, every year (UNEP-FI and PRI, 2010). Whilst the largest of these externalities is the damage impact of climate change, several large externalities (e.g. from freshwater extraction, waste generation, land and sea pollution) appear in the form of losses in public natural capital. The 'public goods' nature of this problem, and the absence of institutions or mechanisms to internalize these externalities, leads many to believe that reforms in micro-level policy might be the only way ahead. Indeed, here there is a growing body of opinion that we need nothing short of a redesign of corporations themselves,[11] as the economy's main agents, if we are to successfully enable a transition to a 'Green Economy'. Among the many changes being sought—including different models of ownership for corporations and changes in finance, advertising, and taxation—an especially important change is that corporations must be responsible for discovering, measuring, and managing their negative externalities down to levels that are acceptable to *stake*holders, not just *share*holders.

'Corporation 20/20', a recent campaign for corporate redesign, sees the process of redesign as an evolutionary one. It argues that corporations, rather like species, evolve by responding to changes in their environment. The operating environment of corporations consists of policies, prices, and institutions, and so the argument of Corporation 20/20 is that exogenous changes are needed in these areas in order to engineer an evolutionary but rapid transformation in the dominant cost-externalizing model that we see today. Corporation 20/20 recommends four agendas for time-bound change which it considers mission-critical for ensuring that economic direction and resource use does not get dangerously close to or rush past planetary boundaries (Rockström et al., 2009). These are: (i) measuring and disclosing externalities; (ii) making advertising more accountable; (iii) limiting leverage for 'too-big-to-fail' corporations; and (iv) replacing profits taxation with taxes on resource extraction and use. Of these four concurrent agendas, three—i.e., changes in the manner in which policies and institutions address externalities (especially those that relate to natural capital), advertising (in that it drives consumer demand and hence resource use), and resource taxation (to the extent that current low levels encourage natural resource extraction)—are relevant to reducing pressures on ecosystem services and biodiversity.

[11] Such as Allen White and Marjorie Kelley's (Tellus Institute) project 'Corporation 20/20'; the recent campaign Corporation 2020 launched at Rio + 20 (<www.corp2020.com>).

The first and perhaps most over-arching change agenda is about measuring, disclosing, and managing down externalities. To take this forward, a 'TEEB for Business Coalition' has been established to bring together global stakeholders to study and standardize methods for natural capital accounting and enable its valuation and reporting in business.[12] This is an area of considerable complexity and challenge, especially the challenge of achieving cohesion across private sector initiatives at different levels, including road-tests and pilot projects by leading corporations, industry-wide initiatives to set guidelines and standards, and over-arching global initiatives such as carbon disclosure, water disclosure, and integrated reporting for corporations. Consistency and comparability of reporting and disclosure have to be achieved at three stages: discovery and quantification of life-cycle impacts on ecosystems for diverse industries and businesses; economic valuation of these impacts using a consistent framework and appropriate industry-wise valuation methodologies; and finally, integrated reporting of all significant impacts, ideally in the form of 'one report'. The many institutional partners of the Coalition, as well as its early movers, have a significant collaboration and coordination challenge ahead to evolve consensus around vision, strategy, and implementation plans.

The TEEB 'community' today includes several hundred economists, ecologists, social scientists, policy-makers, administrators, and business professionals, among others. Quality, transparency, and inclusion have been guiding principles that united them in building this community, and the need for change has been their common driver. Agreeing on a vision and way forward across this community of experts and decision-makers has been perhaps an unstated success of the TEEB project, and one that the recently formed business community of the Coalition may also need to emulate for success in its challenging goal of a global system for measuring and reporting corporate externalities.

7.5 CONCLUDING REMARKS

Valuing nature's services in economic terms is not a political or corporate strategy accepted by everyone. Indeed, the TEEB reports detail both the theory and practice of diverse aspects of the human institution of valuation in

[12] Launched by TEEB Study Leader Pavan Sukhdev, the Coalition's activities focus on global stakeholder engagement, focused research, and development of methods for natural capital accounting. The Coalition's founding members have pioneered much of the science and business case for natural capital valuation and accounting, providing a credible platform to take the business application of this forward.

different social and cultural contexts which are beyond economic considerations. However, it is usually either facile or incorrect to jump from seeking 'valuation' (which can be in the form of value recognition, value demonstration, or value capture supported by appropriate policies and practices) to seeking 'marketization'. Economics is about much more than markets; it is about choices—about using incentives, policies, and regulations; about ensuring access to resources including necessities for healthy living such as clean air and safe water. A broad range of examples cited in the TEEB report suite has shown that successful solutions to biodiversity loss and ecosystem degradation can be devised using economic theory and practice, which are not 'market' solutions as such, although they may use economic argument.

The process of identifying nature's values is not to be taken as an end in itself. It should be treated as a means to better communicate and take account of nature's importance, with particular respect to human well-being. While this is neither necessary nor sufficient to stop all ecosystem degradation and biodiversity loss, it can prove extremely useful if placed in the appropriate context. Valuation can help us rethink our relationship with nature, alerting us to the true consequences of our behaviours and choices.

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Part III

Natural Capital and Accounting

8

Natural Capital

Edward B. Barbier

8.1 INTRODUCTION

An important contribution of natural resource economics has been to treat the natural environment as a form of capital asset, or *natural capital* (e.g. see Clark, 1976; Dasgupta and Heal, 1979; Freeman et al., 1973; Herfindahl and Kneese, 1974). But it has long been argued that the concept of natural capital should not be restricted just to those natural resources, such as minerals, fossil fuels, forests, agricultural land, and fisheries, that supply the raw material and energy inputs to our economies (Freeman et al., 1973; Howe, 1979; Krutilla, 1967; Krutilla and Fisher, 1975; Pearce et al., 1989). Nor should we consider the capacity of the natural environment to assimilate waste and pollution the only valuable 'service' that it performs. Instead, natural capital is much broader, encompassing the whole range of goods and services that the environment provides. Many have long been considered beneficial to humans, such as nature-based recreation, ecotourism, fishing and hunting, wildlife viewing, and enjoyment of nature's beauty. However, natural capital should also comprise those ecosystems that through their natural functioning and habitats provide important goods and services to the economy. Such *ecological capital* is therefore an important component of natural capital (Barbier, 2011*a*; Daily et al., 2000).

In sum, the term 'natural capital' is now frequently employed to define an economy's environment and natural resource endowment—including ecosystems. Humans depend on and use this natural capital for a whole range of important benefits, which are vital to our health, sustenance, and enjoyment of life. For all these reasons, our natural wealth is extremely valuable. But unlike skills, education, machines, tools and other types of human and reproducible capital, we do not have to manufacture and accumulate our endowment of natural assets. Nature has provided this endowment and its benefits to us as part of humankind's common heritage; we have not had to create these assets ourselves.

Yet perhaps because this capital has been endowed, we have tended to view it as limitless, abundant, and always available for our use. The result is that present-day economies have often ended up overexploiting natural capital in the pursuit of economic development, growth, and progress. The unfortunate result is that generations today are leaving too little for future generations to use and benefit from. Over the long term, the consequence is to undermine economic growth and human well-being. Thus, our use of natural capital has significant implications for the overall wealth and sustainable development of an economy.

This chapter will explore these themes by first briefly outlining the early use of the concept of natural capital to describe the environment, through to its more recent extension to include ecosystems. The concept of natural capital has proved useful, as well as controversial, with respect to both the economics of sustainable development and wealth accounting. With respect to sustainable development, the differing perceptions of 'weak' versus 'strong' sustainability hinge on the critical distinction between forms of natural capital that is substitutable and irreversibly depleted. For example, whereas human and reproducible capital may be substitutable for conventional natural resources, such as fossil fuels, land and raw materials, there may be limits on replacing irreversibly lost natural ecosystems and their services. Extending natural capital to include ecosystems also poses challenges for measuring changes in the wealth of an economy, to determine whether it meets the basic sustainability criterion of non-declining welfare.

Already, considerable progress has been made in extending conventionally defined net domestic product (NDP) of an economy to include any appreciation or depreciation in human and various sources of natural capital.[1] In the case of non-renewable resources, such as fossil fuels and minerals, depletion of these resources should be deducted from NDP. For renewable resources, such as forests and fisheries, NDP must include any depreciation (appreciation) in natural resource stocks if current extraction rates are greater (lesser) than biological growth. But, allowances must also be made for changes to ecosystems—or *ecological capital* for short—that affect current economic well-being, either directly or indirectly through supporting production and protecting human lives and property. Accounting for such changes is especially important, given that ecological capital is unlikely to be intact, as many ecosystems continue to be converted to land for economic development and production.

As long as one is careful to account for these direct and indirect contributions of ecological capital to human welfare, then it is possible to extend NDP further to include changes in ecological capital as well. The example of the

[1] See, for example, Aronsson and Löfgren (1996); Arrow et al. (2012); Asheim (1994, 1997); Cairns (2000); Dasgupta (2009); Dasgupta and Mäler (2000); Hamilton and Clemens (1999); Hartwick (1990, 1994); Mäler (1991); Pearce and Atkinson (1993); UNU-IHDP and UNEP (2012); and Weitzman (1976).

USA is used to illustrate the more conventional extension of NDP to include basic changes in human and natural capital. Mangrove ecosystems in Thailand are employed to show the additional extension of adjusting NDP for loss of ecological capital. The next chapter in this book (Hamilton, Chapter 9) shows how NDP can be extended further to account for changes in biodiversity—the range of variation or differences in living organisms found in the environment—which is another important property of ecosystems.

This chapter concludes by exploring additional issues surrounding the concept of natural capital, especially with regard to its practical policy purposes for sustainable economic development and wealth accounting.

8.2 NATURE AS CAPITAL

In order to view the natural environment as a special type of capital asset—a form of 'natural wealth'—then just like any other asset or investment in the economy, the environment must be capable of generating current and future flows of income or benefits. It follows that, in principle, the various components of natural capital can be valued just like any other asset in an economy. Regardless of whether or not there exists a market for the goods and services produced by ecosystems, their social value must equal the discounted net present value (NPV) of these flows.

In the early development of natural resource economics, it became evident that this capital approach applied to certain valuable renewable and natural resource stocks found in the environment, such as mineral ores, energy reserves, fisheries and forests, as stores of wealth (Clark, 1976; Freeman et al., 1973; Dasgupta and Heal, 1979; Herfindahl and Kneese, 1974). But soon there was growing recognition that this concept of 'natural capital' should be extended to other components of the natural environment that also provide valuable flows of goods and services (e.g. see Freeman et al., 1973; Howe, 1979; Krutilla, 1967; Krutilla and Fisher, 1975).

For instance, in the early 1970s, Freeman et al. (1973) proposed that the environment should be considered a 'capital good' for the diverse 'services' that it generates:

> [We] view the environment as an asset or a kind of nonreproducible capital good that produces a stream of various services for man. Services are tangible (such as flows of water or minerals), or functional (such as the removal, dispersion, storage, and degradation of wastes or residuals), or intangible (such as a scenic view). (p. 20)

However, in recent years, there has also been rising concern over the continuing disappearance and degradation of many of the world's ecosystems and the subsequent loss in the many benefits—or 'services'—they provide. This growing literature on ecological services also implies that ecosystems are assets that produce a flow of beneficial goods and services over time.[2] For example, as Daily et al. (2000) state:

> The world's ecosystems are capital assets. If properly managed, they yield a flow of vital services, including the production of goods (such as seafood and timber), life support processes (such as pollination and water purification), and life-fulfilling conditions (such as beauty and serenity). (p. 395)

Ecosystems should therefore be treated as an important asset in an economy, and in principle, ecosystem services should be valued in a similar manner as any form of wealth. That is, regardless of whether or not there exists a market for the goods and services produced by ecosystems, their social value must equal the discounted NPV of these flows.

In sum, the term 'natural capital' denotes an economy's environment and natural resource endowment—including ecosystems—that yields a valuable flow of goods and services to human beings. Although this concept has proved useful in thinking about the environment as a form of wealth, it is not without its controversies. Two issues have especially proven to be difficult to resolve. These are the role of natural capital in sustainable economic development, and accounting for the contributions of changes in natural capital to the overall wealth of an economy.

8.3 SUSTAINABLE DEVELOPMENT

Economic interpretations of sustainability usually take as their starting point the consensus reached by the World Commission on Environment and Development (WCED), which is often referred to as the 'Brundtland Commission' after its chairperson, former Norwegian Prime Minister Gro Harlem Brundtland. Chapter 2 of WCED (1987) defines sustainable development as:

> [d]evelopment that meets the needs of the present without compromising the ability of future generations to meet their own needs.

The Brundtland Commission's definition appeals to economists as it is consistent with a *capital approach* to sustainable development, and thus can also account for natural capital (Pearce et al., 1989). The capital approach to sustainability is summarized schematically in Figure 8.1, and it flows directly from the WCED definition of sustainable development.

[2] See, for example, Barbier (2007, 2011*a*); Daily (1997); Daily et al. (2000); EPA (2009); MA (2005); NRC (2005); Polasky and Segerson (2009); and TEEB (2010).

Economists are generally comfortable with the WCED's broad interpretation of sustainability, as it translates easily into economic terms: an increase in well-being today should not have as its consequence a reduction in well-being tomorrow (see Figure 8.1).[3] That is, future generations should be entitled to at least the same level of economic opportunities—and thus at least the same level of economic welfare—as currently available to present generations. Consequently, economic development today must ensure that future generations are left no worse off than present generations. Or, as some economists have succinctly put it, per-capita welfare should not be declining over time (Arrow et al., 2012; Pezzey, 1989).

As noted in Figure 8.1, it is the *total* stock of capital employed by the economic system, including natural capital, which determines the full range of economic opportunities—and thus well-being—available to both present and future generations. Society must decide how best to use its total capital stock today to increase current economic activities and welfare, and how much it needs to save or even accumulate for tomorrow, and ultimately, for the well-being of future generations.

However, it is not simply the aggregate stock of capital in the economy that may matter but also its composition—in particular, whether present generations are using up one form of capital to meet the needs of today. For example, much of the interest in sustainable development has risen out of concern that current economic development may be leading to rapid accumulation of reproducible and human capital, or *human-made capital*, but at the expense of excessive depletion and degradation of *natural capital*. The major concern has been that, by depleting the world's stock of natural wealth irreversibly, the development path chosen today will have detrimental implications for the well-being of future generations. In other words, according to this view, current economic development is essentially unsustainable.

From an economic standpoint, the critical issue of debate is not whether natural capital is being irreversibly depleted, but what the costs are of these losses and whether society today can compensate future generations for the current loss of natural capital. For example, as Pearce et al. (1989) state,

> future generations should be compensated for reductions in the endowments of resources brought about by the actions of present generations. (p. 3)

[3] As Bishop (1993) maintains, stated in this way the objective of 'sustainability' is different from that of the standard economic goal of 'efficiency'. That is, there are potentially an infinite number of development paths for an economy, only some of which are sustainable. Efficiency therefore does not guarantee sustainability, as some efficient paths are not sustainable. At the same time, there is no reason why an economy could not be both efficient and sustainable.

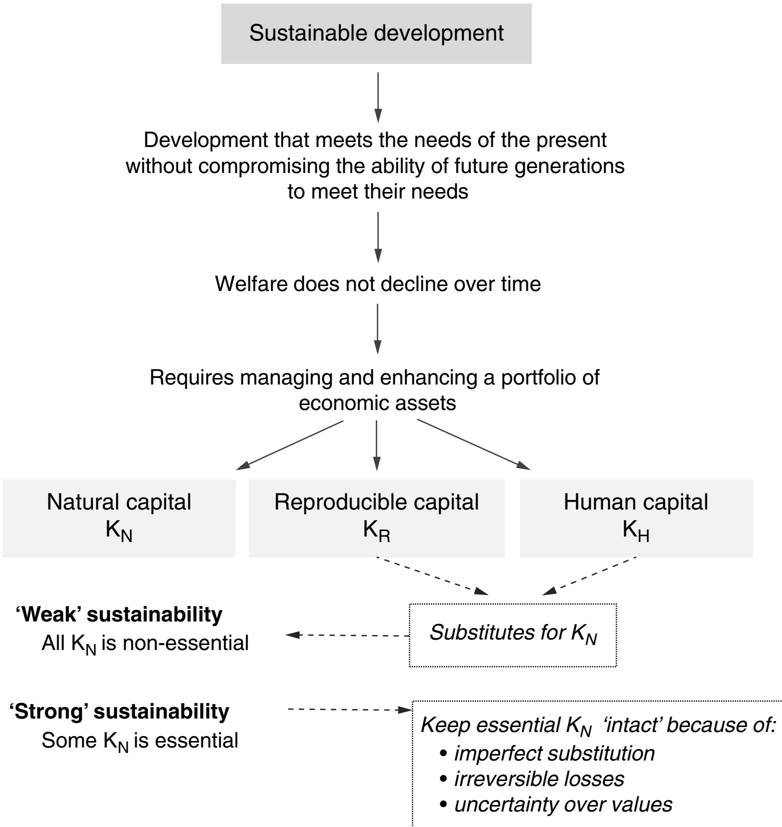

Fig. 8.1. The capital approach to sustainable development

A key question is what form should this compensation take? On this issue, economists diverge in opinion. This difference of view is often referred to as *weak sustainability* versus *strong sustainability* perspectives.

8.4 WEAK AND STRONG SUSTAINABILITY

Although economists generally endorse the capital approach to sustainability, sometimes divisions emerge over the special role of natural capital in sustainable development. The main disagreement is whether or not natural capital has a unique or *essential* role in sustaining human welfare, and thus whether special 'compensation rules' are required to ensure that future generations are not made worse off by natural capital depletion today. These two contrasting views are now generally referred to as *weak sustainability* and *strong sustainability* (see Figure 8.2).[4]

According to the *weak sustainability* view, there is no inherent difference between natural and other forms of capital, and hence the same compensation rules ought to apply to both. As long as the natural capital that is being depleted is replaced with even more valuable reproducible and human capital, then the value of the aggregate stock—comprising human, reproducible, and the remaining natural capital—is increasing over time.[5] Maintaining and enhancing the total stock of all capital alone is sufficient to attain sustainable development.

In contrast, proponents of the *strong sustainability* view argue that reproducible or human capital cannot substitute for all the environmental resources comprising the natural capital stock, or all of the ecological services performed by nature. Consequently, the strong sustainability viewpoint questions whether human, reproducible, and natural capital comprise a single homogeneous total capital stock. Instead, proponents of strong sustainability maintain that some forms of natural capital are *essential* to human welfare, particularly key ecological goods and services, unique environments and natural habitats, and even irreplaceable natural resource attributes, such as biodiversity. Uncertainty over the true value to human welfare of these important assets, in particular the value that future generations may place on them if they become increasingly scarce, further limits our ability to determine whether we can adequately compensate future generations for irreversible losses in such essential natural capital today. Thus the strong sustainability view suggests that environmental resources and ecological goods and services that are essential for human welfare and cannot be easily *substituted* by human and reproducible capital should be protected and not depleted. The only satisfactory compensation rule for protecting the welfare of future generations is to keep essential natural capital intact. That is, maintaining or increasing the value of the total capital stock over time in turn requires keeping the non-substitutable and essential components of natural capital constant over time.

The debate between weak and strong sustainability perspectives is not easy to reconcile. Nevertheless, it is clear that the *minimum* criterion for attaining sustainable economic development is ensuring that an economy satisfies *weak sustainability* conditions. That is, as long as the natural capital that is being

[4] For further discussion of this distinction between sustainability perspectives see Barbier et al. (1994); Howarth and Norgaard (1995); Neumayer (2010); Pearce et al. (1989); Pearce and Barbier (2000); Toman et al. (1995); and Turner (1993).

[5] Note, however, that rapid population growth may imply that the value of the per-capita aggregate capital stock is declining even if the total value stays the same. Moreover, even if the per-capita value of the asset base were maintained, it may not imply non-declining welfare of the majority of people. These considerations also hold for the strong sustainability arguments discussed later.

Weak sustainability	Strong sustainability
• No difference between natural and other capital.	• Cannot view natural, reproducible, and human capital as a homogeneous stock.
• As long as depleted natural capital is replaced with even more valuable reproducible and human capital, then the value of the aggregate stock will increase.	• Cannot always substitute for natural capital, as uncertainty over current and future values of ecological goods and services, unique environments, and biodiversity mean that some natural capital is essential and cannot be replaced.
• Sustainability requires maintaining and enhancing the value of the aggregate capital stock.	• Sustainability requires maintaining and enhancing the value of the aggregate capital stock, and preserving essential natural capital.

Fig. 8.2. Weak and strong sustainability

depleted is replaced with even more valuable reproducible and human capital, then *the value of the aggregate stock*—comprising human, reproducible, and the remaining natural capital—should be increasing over time. This in turn requires that the development path of an economy is governed by certain principles.[6] First, environmental and natural resources must be managed efficiently so that the welfare losses from environmental damages are minimized and any resource rents earned after 'internalizing' environmental externalities are maximized. Second, the rents arising from the depletion of natural capital must be invested into other productive economic assets.

However, the conditions under which depletion of natural capital may or may not lead to more sustainable development clearly depend on what we include as this form of wealth. For example, because they produce goods and services that support economic activity and enhance human welfare, ecosystems should and can be viewed as economic assets. As Dasgupta (2008) maintains, ecosystems are a very unique form of wealth compared with reproducible human-made capital:

> Ecosystems are capital assets. Like reproducible capital assets (roads, buildings, and machinery), ecosystems depreciate if they are misused or are overused. But they differ from reproducible capital assets in three ways: (1) depreciation of

[6] These principles are inspired conceptually by 'Hartwick's rule' (Hartwick, 1977), which is often also referred to as the Hartwick–Solow rule, in recognition that Solow (1974) first derived the principle that reinvestment of the rents generated from the intertemporally efficient use of exhaustible natural resources can be made in reproducible capital in order to ensure a constant stream of consumption over time. Solow (1993) provides an excellent summary of the implications of Hartwick's rule for economic sustainability.

> natural capital is frequently irreversible (or at best the systems take a long time to recover), (2) except in a very limited sense, it isn't possible to replace a depleted or degraded ecosystem by a new one, and (3) ecosystems can collapse abruptly, without much prior warning. (p. 3)

The provision of goods and services by many ecosystems is poorly understood, and their values are often not marketed or even known. In addition, the presence of ecological thresholds and the threat of collapse mean that we are often unaware of the full ecological and economic consequences of current levels of ecosystem degradation and conversion. Moreover, once converted, ecosystems may not be irreversibly lost, but reparation and restoration could be prohibitively expensive, if not technically infeasible in some cases. Improving our knowledge in all of these areas is a critical task. Better understanding of the complex workings of ecosystems and the value of the various goods and services they produce may also help to resolve the weak versus strong sustainability debate over what constitutes essential natural capital.

In sum, even if we believe that some natural ecosystems and unique environments might need to be kept 'intact', much more work still needs to be done in determining how essential is this natural wealth to the welfare of current and future generations, and how costly it may be to protect and conserve such assets. Resolving the weak versus strong sustainability debate does not mean an end to the contribution of economics to environmental policy debate. To the contrary, choices and trade-offs over environmental conservation are still required, and that in turn calls for better analysis of the non-market values of natural capital, understanding the causes and impacts of ongoing environmental degradation, and extending conventionally defined NDP of an economy to include any appreciation or depreciation in human and various sources of natural capital, including ecological capital. It is the latter analysis to which this chapter now turns to illustrate further the importance of the concept of natural assets.

8.5 WEALTH ACCOUNTING AND NATURAL CAPITAL

For most economies, the standard indicator of economic progress is real per-capita gross domestic product (GDP), the market value of all final goods and services produced within the economy. The problem with GDP, however, is that it does not reflect changes in the capital stock underlying the production of goods and services. Since the purpose of new investment is to increase the net quantity and quality of the economy's total capital stock, or wealth, adjusting GDP for net new investment (after depreciation) would measure more accurately whether net additions to capital are occurring. And, as has been demonstrated, economic development is sustained if and only if such investment in overall wealth is non-negative over any time period (Arrow et al., 2012; Dasgupta, 2009; Dasgupta and Mäler, 2000; Hamilton and Withagen, 2007; Hartwick, 1990; Pezzey, 1997).

The idea of deducting any real capital depreciation from GDP to obtain a 'net' domestic product measure is not new. Lindahl (1933) first provided the justification by suggesting that an economy's income should exceed current consumption, including any consumption of existing capital, to prevent comprehensive wealth from declining. However, the total stock of economic assets should be much broader than conventional reproducible (or fixed) assets, such as roads, buildings, machinery, and factories. A growing literature has demonstrated that any system of NDP accounts for an economy should be extended to include two other critical economic assets—human and natural capital. Investments in human capital, such as education and skills training, are essential to sustaining development. Similarly, an economy's endowment of natural resources is an important form of natural wealth. Thus, a better indicator of an economy's progress would be an expanded measure of NDP that is 'adjusted' for real depreciation in reproducible and natural capital, as well as any net additions to human capital (Aronsson and Löfgren, 1996; Arrow et al., 2012; Dasgupta, 2009; Hamilton and Clemens, 1999; Hartwick, 1990).

If ecosystems are also considered capital assets—or *ecological capital*—then efforts to modify NDP to include natural and human capital should account for the contributions of ecosystems as well. However, accounting for depreciation in ecological capital should involve similar rules for estimating the depreciation and appreciation of other assets in an economy. Barbier (2012, 2013) shows that, by adopting and extending the inclusive wealth methodology developed by Dasgupta (2009) and Arrow et al. (2012), it is possible to include ecological capital as well. Such an accounting framework defines the aggregate wealth as the shadow value of the stocks of all the assets of an economy, which should include reproducible, human, and natural capital. Adding in ecological capital is also straightforward, although two important accounting rules emerge (Barbier, 2012, 2013). First, confirming a result initially identified by Mäler (1991) for environmental resources generally, accounting for ecosystems and their services leads to adjusting NDP for the direct benefits provided by the current stock of ecosystems but not for their indirect contributions in terms of protecting or supporting economic activity, property and human lives. Second, as Hartwick (1992) has illustrated in the case of tropical deforestation, when ecosystems are irreversibly converted for economic development, NDP must be further modified to reflect any capital revaluation that occurs with the current conversion of ecological capital to other land uses.

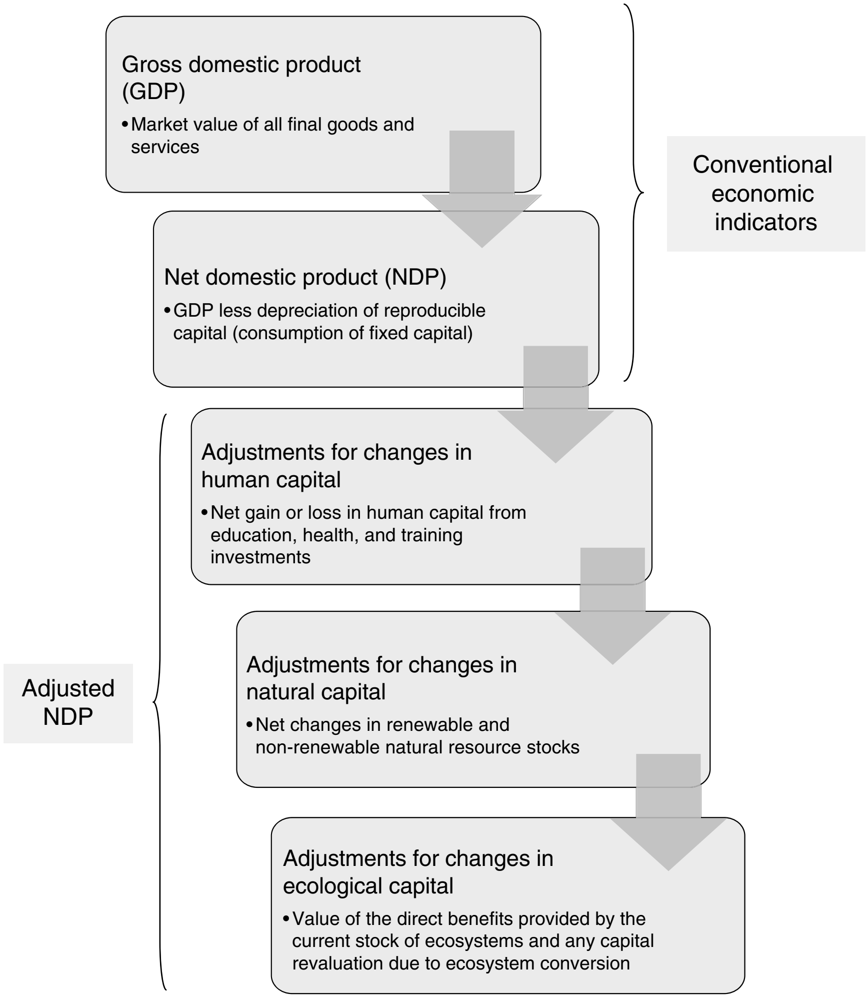

Fig. 8.3. Adjusting NDP for reproducible, human, and natural capital

Figure 8.3 outlines the basic methodology required to adjust NDP for reproducible, human, and natural capital, including ecological capital. Once the changes in the value of these stocks are accounted for, the result is adjusted NDP.

As indicated in Figure 8.3, GDP and NDP are conventional indicators that are regularly reported in the national accounts for most economies. As discussed earlier, however, NDP accounts for the 'depreciation' in value of only reproducible capital. Instead, as outlined in Figure 8.3, additional adjustments for changes in human, natural, and ecological capital are required to determine whether current production in the economy is reliant on depreciating or adding to overall wealth. First, any current investments in education, training, and health are likely to lead to net gains in human capital. Second, NDP needs to be adjusted for the depletion of non-renewable resources, such as fossil fuels and minerals; for renewable resources, such as forests and fish, NDP must include any net gains or losses in these stocks depending on whether depletion exceeds biological growth. Finally, NDP should be adjusted for the direct benefits provided by the current stock of ecosystems as well as any capital revaluation that occurs if ecosystems are converted to other land uses.[7]

To illustrate how these accounting rules for adjusting NDP are applied, two country cases are examined. The example of the USA economy is first employed to show the basic adjustments to include changes in human and natural capital. Mangrove ecosystems in Thailand are then used to demonstrate the additional extension of adjusting NDP for loss of ecological capital.

8.5.1 USA

An approximate estimate of adjusted net domestic product (ANDP) per capita along the lines suggested by Figure 8.3 can easily be constructed for the USA from the World Bank's *World Development Indicators* (World Bank, 2011). This dataset includes consumption of fixed capital, total education expenditures, and depreciation of some natural resources such as fossil fuels, minerals, and timber from 1970 to 2008 for many economies.

As shown by Barbier (2011*b*), these data can be used to construct a measure of ANDP for the USA in the following manner. First, by deducting consumption of fixed capital investment from GDP, conventionally defined NDP is obtained. Second, by using education expenditures as a proxy for net gains in human capital, and mineral and energy depletion as an adjustment for depreciation of non-renewable natural capital, ANDP is estimated. Unfortunately, there is insufficient data to allow the additional adjustment of ANDP to measure net gains or losses in renewable natural capital or ecological capital.

Figure 8.4 compares the resulting trends in real GDP and ANDP per capita (constant 2000 US$) for the United States from 1970 to 2008. Although the two measures generally follow the same long-run trend, ANDP per capita is consistently lower than GDP per capita. In addition, the gap between the two indicators has been widening. In 1970 real GDP per capita was $18,229, and

[7] As Hamilton and Clemens (1999) have pointed out, if the direct benefits of any ecosystem or 'environmental' services are negatively affected by the accumulation of pollution, then one should also account for the net changes in this harmful 'stock' in the environment. Arrow et al. (2012) and Dasgupta (2009) apply similar reasoning to account for the climate-related damages caused by the accumulation of greenhouse gas emissions.

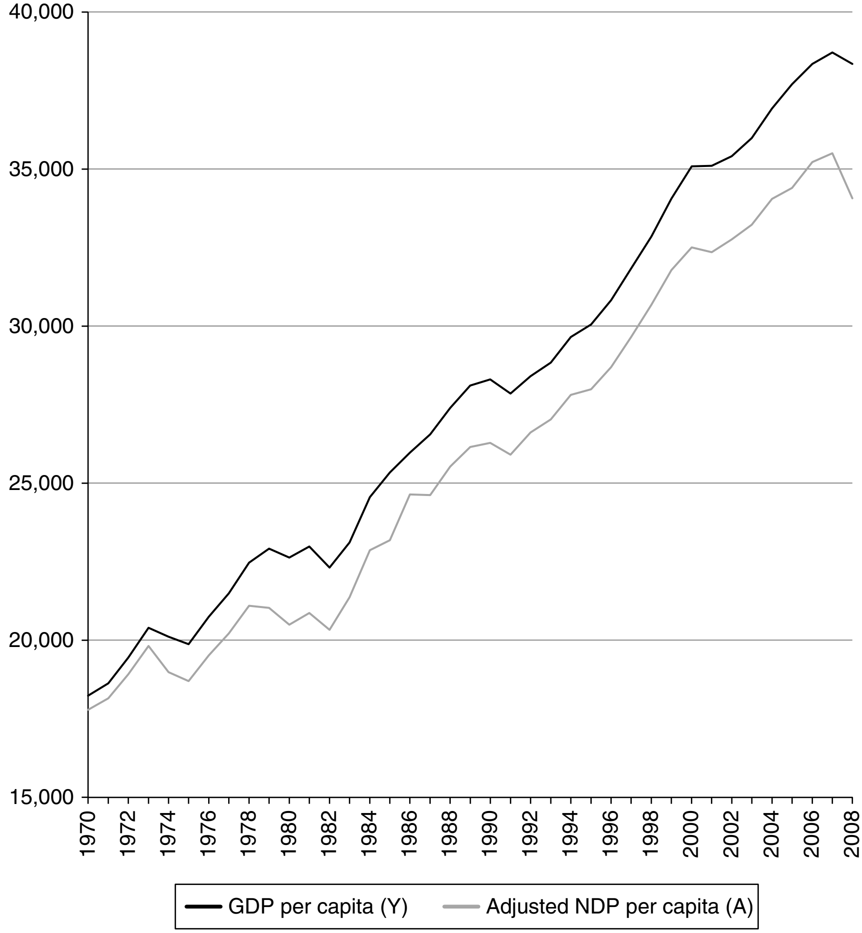

Fig. 8.4. Real GDP and ANDP per-capita trends for the USA, 1970–2008 (constant 2000 US$)

Notes: ANDP is adjusted net domestic product, or GDP less consumption of fixed capital and natural resource depletion, plus education expenditure.

Energy depletion is the ratio of the value of the stock of energy resources to the remaining reserve lifetime (capped at twenty-five years). It covers coal, crude oil, and natural gas. Mineral depletion is the ratio of the value of the stock of mineral resources to the remaining reserve lifetime (capped at twenty-five years). It covers tin, gold, lead, zinc, iron, copper, nickel, silver, bauxite, and phosphate.

Source: Adapted from Barbier (2011*b*, Figure 1). Data from World Bank (2011).

ANDP per capita was $17,786, but by 1990, GDP per capita had risen to $28,299, whereas ANDP per capita was $26,288. By the 2000s, the gap had increased further; by 2007, real GDP per capita reached $38,701, and ANDP per capita was only $35,497. Both indicators fell in 2008, signalling the start of the Great Recession. However, the decline in ANDP per capita of 4.0 per cent over 2007–8 was significantly greater than the 0.9 per cent fall in GDP per capita.

Table 8.1. Changes in real GDP and ANDP per capita for the USA, 1970–2008 (%)

Average annual growth rate per capita (constant 2000 US$)					
	GDP	ANDP	Consumption of fixed capital	Education expenditures	Energy and mineral depletion
1970–79	2.4	1.7	4.8	1.5	20.3
1980–89	2.7	3.0	1.7	−1.9	−24.8
1990–99	2.2	2.2	2.2	1.4	−10.7
2000–08	1.4	1.1	2.6	1.3	16.9
1970–2008	2.0	1.9	2.6	0.5	−1.5

Notes: ANDP is adjusted net domestic product, or GDP less consumption of fixed capital and natural resource depletion, plus education expenditure.

Energy depletion is the ratio of the value of the stock of energy resources to the remaining reserve lifetime (capped at twenty-five years). It covers coal, crude oil, and natural gas. Mineral depletion is the ratio of the value of the stock of mineral resources to the remaining reserve lifetime (capped at twenty-five years). It covers tin, gold, lead, zinc, iron, copper, nickel, silver, bauxite, and phosphate.

Source: Adapted from Barbier (2011*b*, Table 1). Data from World Bank (2011).

Further insights from these trends can be gained from examining changes in the various components of ANDP per capita. Table 8.1 indicates that, from 1970 to 2008, the average annual growth rates in real GDP per capita (2.0 per cent) and real ANDP per capita (1.9 per cent) were similar. Fixed capital depreciation per capita grew 2.6 per cent annually, but energy and mineral depletion per person fell by 1.5 per cent annually. Educational expenditures per capita grew modestly each year by 0.5 per cent.

However, for each decade, annual average growth rates vary significantly. Of particular concern is that the pattern of growth of the 1970s is being replicated in the 2000s. In the 1970s, per capita reproducible and natural capital depreciation rose substantially each year (4.8 per cent and 20.3 per cent, respectively), whereas per capita educational expenditures grew only modestly (1.5 per cent). As a result, average annual growth in ANDP per capita lagged behind growth in GDP. Although energy and mineral depletion fell over subsequent decades, from 2000 to 2008 natural resource depreciation per capita grew 16.9 per cent annually. The average annual growth rate in fixed capital consumption was also higher (2.6 per cent), whereas the annual average growth in educational expenditures per capita was lower (1.3 per cent). Once again, growth in ANDP per capita (1.1 per cent) lagged behind growth in GDP per capita (1.4 per cent).

These comparisons of GDP and ANDP per capita for the USA are revealing in several respects. First, ANDP is a better indicator of whether or not current increases in an economy's real income from domestic production are leading to net additions to capital. Second, the US economy remains dependent on depreciating its mineral and energy assets. Reducing this dependence through clean energy investments is not just an urgent priority in environmental and energy security terms, it may be an economic necessity. Finally, the trends indicate that overall net investment growth is starting to lag in the US economy, and given the economic impacts of the Great Recession, since 2008 the problem has worsened. If trends since 2000 are not reversed soon, then the US economy looks increasingly less sustainable than ever.

8.5.2 Thailand

The previous example of the USA shows how accounting for changes in natural capital, along with reproducible and human capital, is important for determining whether current production is adding to or depleting an economy's current stock of wealth. However, for lack of data, changes in ecological capital were not included in the US example. The purpose of this section is to draw on the example of mangrove loss in Thailand to illustrate how the ANDP methodology outlined in Figure 8.3 can also account for ecosystem change. Based on Barbier (2012, 2013), the case study illustrates the two adjustments to NDP due to ecological capital: the value of the *direct benefits* provided by the current stock of ecosystems, and any capital revaluation that occurs as a result of ecosystem conversion to other land uses. Barbier (2013) shows how NDP adjustments also could take into account the risk of ecological collapse to mangrove systems in Thailand from extensive land conversion, and the next chapter (Hamilton, Chapter 9) shows how the contribution to NDP of biodiversity in Thailand and other countries can be approximated through using the value of protected areas.

Thailand is estimated to have lost around a third of its mangroves since the 1960s, mainly to shrimp farming expansion and other coastal development (FAO, 2007*a*; Spalding et al., 2010). During this period, real GDP per capita in Thailand has increased fivefold (World Bank, 2011). A measure of the adjusted NDP, taking into account human and natural capital loss since 1970, is constructed. Based on estimates of four mangrove ecosystem benefits—collected products, habitat–fishery linkages, storm protection, and carbon sequestration—the methodology of adjusting NDP for the value of ecosystems is also included as an illustration.

Thailand is estimated to have had around 368,000 hectares (ha) of mangroves in 1961 (FAO, 2007*b*; Spalding et al., 2010). Mangrove deforestation proceeded swiftly in the 1970s and 1980s, but since 2000, the area of mangroves seems to have stabilized around 240,000 to 250,000 ha (FAO, 2007*b*; Spalding et al., 2010). The main cause of mangrove loss in Thailand is attributed to conversion to shrimp aquaculture (Aksornkoae and Tokrisna, 2004). The main reason for the slowdown in mangrove loss is that many of the suitable sites for establishing shrimp farms in the Gulf of Thailand have been deforested, whereas the mangrove areas on the Andaman Sea (Indian Ocean) coast are too remote and less suitable for shrimp farms (Barbier and Cox, 2004).

The valuation estimates that are used for accounting for the current benefits of mangroves as well as their capitalized values for Thailand over 1970 to 2009 are described in detail in Barbier (2012, 2013). The four principal ecosystem goods and services are the role of mangroves as natural 'barriers' to periodic damaging coastal storm events, their role as nursery and breeding habitats for offshore fisheries, their ability to store carbon, and the exploitation of mangrove forests by coastal communities for a variety of wood and non-wood products. As outlined in Barbier (2012, 2013), these four benefits of mangroves in Thailand have a constant 2000 US$ capitalized value of $21,443 per ha. As the main activity responsible for mangrove conversion in Thailand has been shrimp aquaculture, the capitalized value (in 2000 US$) of this alternative use of mangrove ecosystems is $1,351 per ha. Note that, because the capitalized value, or 'price', of mangroves converted to shrimp farming is less than the capitalized value of mangroves, the NDP of Thailand should be adjusted for this depreciation in mangrove capital.

However, not all the current benefits of mangroves impact welfare directly, but may do so only through support or protection of economic activity and property. That is certainly the case for storm protection benefits of mangroves, which are estimated through an expected damage approach that determines their value in terms of protecting economic property (Barbier, 2007). As this benefit is already accounted for in the current market values of property, to avoid double-counting, the NDP of the Thai economy should not be adjusted to include the benefit of storm protection provided by the current stock of mangroves. Similarly, a survey of four Thai villages from two coastal provinces indicates that only 12.4 per cent of the value of collected wood and non-wood products from mangroves, and 5.3 per cent of the value of coastal fishery harvests, can be attributed to subsistence production (Sarntisart and Sathirathai, 2004).[8] Thus, the NDP should be adjusted only for these subsistence contributions of these two benefits of the mangroves in Thailand.

Using the data from Barbier (2012), Table 8.2 depicts the per capita wealth accounting estimates for Thailand's mangroves from 1970 to 2009. Average annual mangrove loss has fallen steadily in every decade since the 1970s (see also FAO, 2007*b*; Spalding et al., 2010). Nevertheless, because around a third of the mangrove area has been deforested from 1970 to 2009, whereas Thailand's population has nearly doubled over this period, the value of current per capita benefits of mangroves has halved since the 1970s, from $0.57 to $0.28 per person.[9] In the 1970s, when mangrove loss in Thailand was at its highest, mangrove depreciation amounted to $2.26 per person, whereas by the 2000s, it

[8] The four villages were Ban Sam Chong Tai and Ban Bang Pat of Phang-nga Province, and Ban Gong Khong and Ban Bkhlong Khut in Nakhon Si Thammarat Province.

[9] According to World Bank (2011), in 1970 Thailand's population was 36.9 million and grew steadily to 68.7 million by 2009.

had fallen to only \$0.03 per capita. The result is that the net value of mangroves per capita in Thailand, which is the total value less mangrove depreciation, was actually negative in the 1970s and 1980s, averaging –\$1.69 and –\$0.76 per person, respectively. However, in the 1990s and 2000s, the net value was slightly positive, averaging \$0.11 and \$0.22, respectively.

Table 8.3 depicts an approximate estimate of ANDP per capita for real changes in reproducible, human, and natural capital for Thailand over 1970 to 2009. ANDP is GDP less consumption of fixed capital and natural resource depletion, plus education expenditure and net values of mangrove depletion. The latter estimate is based on the net value of mangroves from Table 8.2. Since the 1970s, both consumption of fixed capital and natural resource depreciation have increased significantly in Thailand. The value of expanding human capital, as proxied by education expenditures, has also increased, and because of the slowdown in mangrove loss, the net value of this ecological capital has gone from a negative to a positive contribution to NDP. Overall, the value of mangroves and expanding human capital has not kept pace with reproducible capital depreciation and natural resource depletion in Thailand. As a consequence, adjusted NDP per capita in Thailand has remained consistently below GDP per capita since the 1970s.

The trends in real GDP and ANDP per capita for Thailand from 1970 to 2009 are depicted in Figure 8.5. As shown in the figure, since 1990 the gap between GDP and ANDP per capita in Thailand has widened significantly. As in the case of the USA, this raises concerns about the lag in net investment growth in the Thailand economy, and its implications for future sustainability.

To summarize, because many of the benefits provided by the current stock of mangroves in Thailand arise through supporting or protecting marketed production and property, these benefits should already be included in the GDP estimates for Thailand. However, any adjusted NDP measure does need to take into account the current direct benefits provided by mangroves in the form of carbon sequestration, habitat, and breeding ground services that support any fishery harvests consumed by coastal households and mangrove products that also comprise subsistence consumption. On the other hand, all future mangrove benefits are lost as a result of mangrove conversion, which has been substantial in Thailand since the 1970s. The substantial mangrove depreciation that occurred in the 1970s and 1980s meant that the net value of mangroves was actually negative in these decades. Although mangrove deforestation and thus its capital depreciation has slowed since, the net value of mangroves per capita, as an indicator of its contribution to the wealth of Thailand, is still extremely low. Thus, the Thailand mangrove case study not only provides an illustration of the adjusted NDP methodology for ecological capital, but also illustrates how significant loss of this capital can influence its net value in wealth accounts.

Table 8.2. Wealth accounting for mangrove capital, Thailand 1970–2009

	Average annual mangrove loss (ha)	Average annual values per capita (constant 2000 US$)					Mangrove depreciation	Net value of mangroves
		Storm protection	Habitat–fishery linkage	Wood and non-wood products	Carbon sequestration	Total value of mangroves		
1970–79	4,676	–	0.11	0.10	0.36	0.57	2.26	−1.69
1980–89	2,980	–	0.08	0.07	0.25	0.40	1.16	−0.76
1990–99	610	–	0.06	0.06	0.20	0.32	0.21	0.11
2000–09	97	–	0.05	0.05	0.18	0.28	0.03	0.25

Notes: As storm protection value is based on expected damages to economic property, it is assumed that this benefit is already accounted for in the current market values of property. Current habitat–fishery linkages benefits are based only on the imputed subsistence value, which, based on a survey of four Thai coastal villages, is approximately 5.3% of total household income (Sarntisart and Sathirathai, 2004, Tables 6.3 and 6.4). Current wood and non-wood product benefits are based only on the imputed subsistence value, which, based on the survey of four villages, is approximately 12.4% of total household income (Sarntisart and Sathirathai, 2004, Tables 6.3 and 6.4).

Source: Based on Barbier (2012, Table 2).

Table 8.3. Real GDP and ANDP per capita, Thailand, 1970–2009

	Average annual values per capita (constant 2000 US$)					
	GDP	ANDP	Consumption of fixed capital	Natural resource depletion	Education expenditure	Net value of mangroves
1970–79	617	544	89	13	30	−1.7
1980–89	956	852	130	19	46	−0.8
1990–99	1,793	1,563	296	20	86	0.1
2000–09	2,291	2,041	280	79	109	0.3

Notes: ANDP is adjusted net domestic product, or GDP less consumption of fixed capital and natural resource depletion, plus education expenditure and net value of mangroves (estimated in Table 8.2).

Natural resource depletion is the sum of net forest depletion, energy depletion, and mineral depletion. Net forest depletion is unit resource rents times the excess of roundwood harvest over natural growth. Energy depletion is the ratio of the value of the stock of energy resources to the remaining reserve lifetime (capped at twenty-five years). It covers coal, crude oil, and natural gas. Mineral depletion is the ratio of the value of the stock of mineral resources to the remaining reserve lifetime (capped at twenty-five years). It covers tin, gold, lead, zinc, iron, copper, nickel, silver, bauxite, and phosphate.

Source: Barbier (2012, Table 3). Data from World Bank (2011), except for net value of mangroves, which is from Table 8.2 of this chapter.

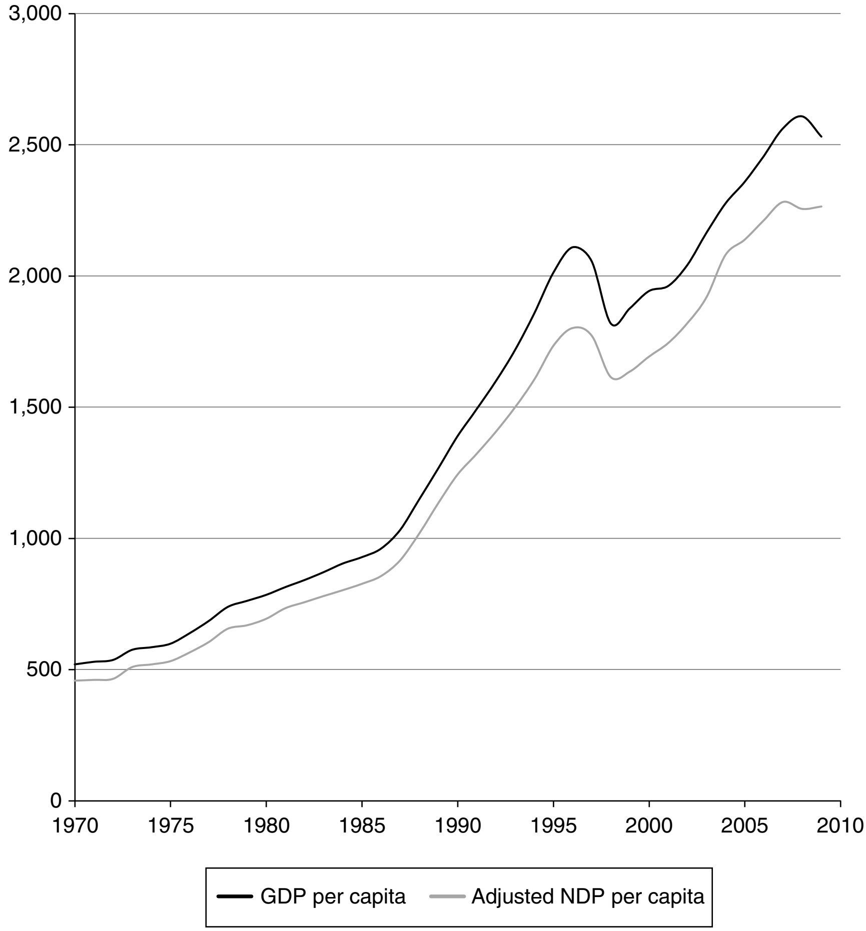

Fig. 8.5. Real GDP and ANDP per capita trends for Thailand, 1970–2009 (constant 2000 US$)

Notes: ANDP = adjusted net domestic product, or GDP less consumption of fixed capital and natural resource depletion, plus education expenditure and net value of mangroves

Source: Adapted from Barbier (2012, Figure 4).

Finally, because the substantial conversion of mangroves increases the risk of collapse of the remaining habitat, NDP should be adjusted further to account for this possible outcome; Barbier (2013) shows how this adjustment to NDP for the risk of collapse to Thailand's mangroves should occur. In addition, an important property of mangroves and other natural habitat is biological diversity, the range of variation or differences in living organisms found within ecosystems. Chapter 9 discusses the various methods by which the value of biodiversity can be accounted for in NDP, and uses the example of Thailand as well as other countries to illustrate one approach that uses the value of protected area as an approximation.

8.6 CONCLUSION

Sustainable development has now become widely accepted as an essential economic goal. The capital approach to sustainability has also helped to clarify the implications for economic policy. The role of policy is to determine how much of an economy's total capital stock today should be used to increase current economic activities and welfare, and how much should be saved or even accumulated for the benefit of future generations.

However, an important advance in economic thinking has been to recognize that the total stock of economic assets essential to human welfare should be much broader than conventional reproducible (or fixed) assets, such as roads, buildings, machinery, and factories. Although it has long been recognized that investments in human capital, such as education and skills training, are also essential to sustaining development, it is increasingly accepted that an economy's endowment of natural resources is also an important form of natural wealth.

When the concept of natural capital is further extended to include more complex environmental assets such as ecosystems, then it can become difficult to determine the most efficient and sustainable trade-offs between environmental conservation and development. Physical or human capital may simply not be good substitutes for all the environmental resources comprising the natural capital stock, or all of the ecological services performed by nature. This means that economists, working with ecologists and other natural scientists, still need to determine the extent to which certain ecosystems and their services are essential to the welfare of current and future generations, and how costly it may be to protect and conserve this natural wealth. It may mean that assessment of trade-offs between human-made and natural capital should be extended to include environmental degradation that threatens biological productivity, biodiversity, and resilience.

In many cases, however, the transition to more sustainable economic development involves more straightforward improvements in the efficient management of natural resource depletion, pollution, and environmental degradation. If the welfare losses from environmental damages are minimized, and the resource rents earned from more efficient depletion of natural capital are invested in other productive economic assets, then much of the current unsustainable economic activity will disappear.

Accounting for the creation or depletion of economic wealth is therefore a crucial indicator of the degree of sustainability of our current economic activities. One indicator that approximates the extent to which current production is adding to or decimating economic wealth is the NDP of an economy, provided that this indicator accounts for the depreciation of all forms of capital—reproducible, human, and natural capital. As shown by the example of the USA included in this chapter, by using education expenditures as a proxy for net gains in human capital, and mineral and energy depletion as an adjustment for depreciation of non-renewable natural capital, such a measure of adjusted NDP—or ANDP—can be easily estimated. The example of mangrove loss in Thailand further illustrates how the ANDP indicator can be extended to account for changes in ecological capital. Such estimates demonstrate that the concept of natural capital and its relationship to the overall wealth of an economy can be translated into an actual indicator of whether or not current increases in an economy's real income from domestic production are leading to increased sustainability.

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9

Biodiversity and National Accounting

*Kirk Hamilton**

9.1 INTRODUCTION

The absence of any valuation of the depletion and degradation of natural resources and the natural environment in the System of National Accounts (SNA) leads at best to policy complacency and at worst to policy mistakes. As *The Changing Wealth of Nations* (World Bank, 2011) documents, the effect of this absence is likely to be felt most keenly in developing countries, where natural resources and the natural environment constitute 21 per cent to 35 per cent of total wealth.

Ignoring the consumption of all forms of capital flatters the growth rates of gross domestic product (GDP) in all countries, which can be a powerful source of complacency. In poor countries that are highly dependent on exhaustible natural resources, the combination of fiscal policies and resource sector policies can result in gross wealth creation that is positive and growing, while the net change in produced plus natural wealth, unmeasured in the SNA, is negative—this is a policy mistake of the first order.

As a large literature attests, and as will be reviewed later, biodiversity is a source of economic benefits. Most of these benefits are currently measured in the SNA. The problem, and a key source of policy errors, is that there is no *explicit* measure of these benefits. As a result, development decisions guided by the traditional indicators of the SNA, such as GDP growth, can harm natural areas and the biodiversity which they harbour, leading to unforeseen losses of well-being. Policy surprises abound.

For commercial natural resources the adoption of the System of Environmental-Economic Accounting (SEEA) as a United Nations statistical standard

* The opinions expressed are those of the author and not necessarily those of the World Bank Group. The comments of Dieter Helm, Colin Mayer, Ed Barbier and Giles Atkinson are acknowledged with thanks. The usual caveats apply.

(United Nations, 2012) is a major step forward in defining the contribution of natural resources to national income, including the effects of depleting these resources. From a biodiversity perspective this work is still incomplete, however, because work on accounting for ecosystem services, in particular, is ongoing.

This chapter explores the extent to which biodiversity and changes in biodiversity could be measured within the existing structure of the SNA, as well as assessing how the SNA could be extended to include a broader portion of the value of biodiversity. We begin by summarizing the treatment of natural resources in the national balance sheet account, and review key portions of the literature addressing the economic value of biodiversity. We then turn to a discussion of the accounting issues and potential ways forward in accounting for biodiversity in the SNA.

A significant proportion of the world's biodiversity resides in protected areas. We conclude with an empirical analysis of the value of protected areas as published in *The Changing Wealth of Nations*. A key insight is that there is an opportunity cost that countries face in establishing protected areas. The striking empirical finding is that the annual opportunity cost of maintaining protected areas is a much higher share of GDP in many developing countries than in OECD countries—the opportunity cost exceeds 1 per cent of GDP in fifty-eight developing countries vs. only four OECD countries. And flows of concessional finance to offset this cost, via the Global Environment Facility and its co-finance, average only 8 per cent of the annual opportunity cost in low-income countries.

9.2 BALANCE SHEETS AND BIODIVERSITY

Although it is not generally emphasized when graduate students in economics are taught about national accounting, the national balance sheet is arguably the root construct of the whole SNA. The balance sheet defines the system boundary for the SNA and one of the principal aggregates defined in the SNA, gross national income (GNI), represents the gross return on the assets measured in the balance sheet. In exploring the current and potential treatment of biodiversity in national accounting, therefore, the balance sheet is the appropriate place to start.

Natural resources appear in national balance sheet accounts as a category of non-produced assets. By defining biodiversity to be a property of a natural system, we can begin to understand its role in the SNA. Specifically, SNA 2008 (the latest standard for national accounting, published in United Nations, 2009) characterizes natural assets in the following manner:

> Only those naturally occurring resources over which ownership rights have been established and are effectively enforced can . . . qualify as economic assets and be

> recorded in balance sheets. They do not necessarily have to be owned by individual units, and may be owned collectively by groups of units or by governments on behalf of entire communities. (para 10.167)
>
> . . . In order to comply with the general definition of an economic asset, natural assets must not only be owned but must also be capable of bringing economic benefits to their owners, given the technology, scientific knowledge, economic infrastructure, available resources and set of relative prices prevailing on the dates to which the balance sheet relates. (para 10.168)
>
> . . . When . . . forests or the animals, birds, fish, etc. [living in the wild] are actually owned by institutional units and are a source of benefit to their owners, they constitute economic assets. (para 10.169)

This characterization leans heavily towards commercial natural resources—but, since governments are typically the owners of undeveloped land including protected areas, the door is at least partially open to the inclusion of biodiverse natural areas as economic assets in the balance sheet, to the extent that they yield economic benefits to governments.[1] The obvious next question, of course, is to understand how biodiversity can yield economic benefits, the subject of the next section.

9.3 BIODIVERSITY AS A SOURCE OF VALUE

Polasky et al. (2005) provide a reasonably comprehensive assessment of the economics of biodiversity, and they derive a useful classification of the sources of value derived from biodiversity. First, individual species may have *use* and *existence values*.[2] Examples of the former include hunting, fishing, wildlife photography, and nature tourism. 'Using' species generally entails associated commercial activities which are measured in the SNA, while existence values are not (nor indeed are option values for preserving species). Use values also generate fee income for governments in the form of admission fees to natural areas and licences for the use of particular species.

Biodiversity may also be a source of *bioprospecting* revenues and is an integral part of the *production of ecosystem services* in natural areas.[3] Bioprospecting involves the search for commercially valuable products from natural species, and their discovery yields both commercial activities and a potential

[1] However, it is important to note that the treatment to date in balance sheet accounts has been to value public lands, and their associated biodiversity, at zero.

[2] In this chapter we focus on existence or bequest values linked to biodiversity, but of course these make up part of the broader class of non-use values in the usual Total Economic Value hierarchy.

[3] In fact Polasky et al. (2005) conclude that there is considerable evidence that higher biodiversity is linked to higher productivity of natural areas.

stream of rents to the owners of the species in question. Ecosystem services are broad, covering the categories of supporting, provisioning, regulating, and cultural services as defined in the Millennium Ecosystem Assessment (2005)—provisioning services overlap substantially with the use values just defined.

Heal (2000) identifies two additional sources of value from biodiversity: *knowledge* and *insurance*. Studying species which make up biodiverse communities may yield knowledge of value in the production of pharmaceuticals and the products of biotechnology—this is obviously closely linked to bioprospecting. And the genetic make-up of related species (e.g. wild variants of commercial crops) may provide insurance in the form of new genetic material which can help commercial species such as food crops adapt to pathogens.

Mace et al. (2012) argue that biodiversity can be an important regulator of ecosystem processes, a final ecosystem service, and a good in and of itself. This conception of biodiversity can be mapped to the values already outlined: the regulation of ecosystem processes contributes to the production of ecosystem services; biodiversity as an ecosystem service overlaps with bioprospecting and knowledge values; and biodiversity as a good (charismatic species, for example) is directly linked to use values.

There is a parallel between the insurance values provided by biodiversity and the ecological concept of resilience. Walker et al. (2010) argue that ecosystem resilience is itself an asset that should be valued in any inclusive measure of total wealth. An alternative way to express the argument is to say that expected wealth is positively related to the level of resilience of the ecosystem. To the extent that biodiversity contributes to ecosystem resilience, it can serve as a type of insurance on existing asset values.

It is important to note that at least two of these sources of value for biodiversity may constitute global public goods—some ecosystem services may provide global benefits, by sequestering carbon for example, while knowledge has inherent global public good characteristics (at least in the absence of patents). This is an important consideration since much of the world's biodiversity resides in developing countries.

9.4 POTENTIAL TREATMENTS OF BIODIVERSITY IN THE NATIONAL BALANCE SHEET

While biodiversity has tended to be measured in terms of relative species abundance or joint dissimilarity of species (Polasky et al., 2005), from the perspective of the national balance sheet it is perhaps simplest to conceive of it as a property of a natural area—one property among many, including soil, hydrology, geology, topography, climate, and location. In what follows we concentrate on valuing natural areas conceived of as land[4] possessing a bundle of properties, including biodiversity.

Based on the preceding assessment of biodiversity as a source of value, several potential treatments of biodiversity in the balance sheet accounts suggest themselves. The most direct link concerns use values, where fees paid for the use of nature represent economic benefits accruing to the owners of natural areas and their associated biodiversity. Taking present values of these fees would give a value for natural assets which could fit consistently within the SNA balance sheet account. However, this would not necessarily be a good measure of the economic value of the natural asset because park fees and other usage fees are generally not tapping the full willingness to pay of the people visiting and using the natural area. Additionally, it is important to note that the total willingness to pay to use the natural area is linked to the bundle of properties possessed by the area, including its biodiversity.

In countries where surveys of users have measured the willingness to pay (per person per day)[5] for the enjoyment of the natural area, it may be possible to reflect this in the balance sheet. While the SNA measures only the value of actual transactions, with few imputations, bringing values of natural areas into the balance sheet based on stated willingness to pay could be done if (i) due care is taken in the survey design; and (ii) there is evidence that willingness to pay actually varies as the properties of the natural area, including biodiversity, change. The result would be an imputed value of national wealth and an associated measure of national income which includes the imputed consumption of benefits over and above the actual user fee paid.

Ecotourism is another example of use value which provides economic benefits to natural tourism operators as well as park entry fees to government. If the natural areas visited by the ecotourists are privately owned then the capitalized rents generated by ecotourism already form part of the national balance sheet—although the land may retain its 'natural' characteristics, it is effectively a commercial natural resource. If the ecotourist visits a lodge in a national park, then the rents paid by the lodge owner to the government contribute to government income and therefore to the asset value of the national park.

[4] Marine protected areas lying within a country's exclusive economic zone can have similar values to those we describe, but there may be issues concerning fugitive species passing into and out of national waters.

[5] The stipulation that the surveys measure willingness to pay to use the natural area per person per day is important, since this is in effect a price, and the relevant quantity is the number of person-days demanded. Imputing the additional consumption of benefits from the natural area is therefore consistent with national accounting conventions which stipulate that final expenditure is the summing up of 'p times q' values, excluding consumer surplus. This suggests, however, that the survey should attempt to measure the demand curve for person-days of use of the natural area.

Bioprospecting rents are also economic benefits accruing to the owner of a natural area, but Polasky et al. (2005) note that the expected values of bioprospecting rents are low—generally less than the financial costs of the conservation of natural areas. The value of the knowledge which can be gained from diverse species can in principle be measured and ascribed to the natural areas which are the source of the species.

As highlighted earlier, biodiversity is an integral part of the production of ecosystem services, and there is evidence that increasing biodiversity increases the productivity of natural areas. This suggests that natural areas, including protected areas, can also be valued on the basis of the ecosystem services they provide.

The key point to recognize when valuing ecosystem services is that the great majority of these services are provided to the rest of the economy as externalities[6]—think of pollinators inhabiting a natural area which provide pollination services to surrounding farms. This has two consequences. First, the values of ecosystem services will already be capitalized in other asset values, such as farmland. Second, accounting for ecosystem service values violates the SNA convention that natural resources have value to the extent that they yield economic benefits to their owners—in this case the beneficiaries are third parties, such as owners of farmland who benefit from the external supply of (costless) environmental services.

Adding up the values of the ecosystem services provided by a natural area and capitalizing these values separately in the national balance sheet would therefore typically be double-counting. To the extent that the value of natural areas is expressed through externalities, the role of accounting for ecosystem services is to disaggregate the existing asset values in the national balance sheet into their different sources of value, including sources from the ecosystem services generated by natural areas—total national wealth will not increase.

If we assume that the different use values, bioprospecting/knowledge values and ecosystem service values of biodiverse areas are *in principle* measured in the national balance sheet, an interesting question is whether there are other sources of value from biodiversity which would not already be measured in the values of existing assets. Two possibilities suggest themselves: insurance values and existence values.

Insurance values linked to the biodiversity harboured in natural areas are expected values, based on the probability of a catastrophic risk to a commercial crop (for example), the probability that the natural area contains species which can yield genetic information of use in adapting the commercial crop to withstand the risk, and the value of the potential catastrophe. Assuming the

[6] Perrings (2012) also makes this point, although he focuses on transboundary externalities. Payments for environmental service (PES) schemes effectively internalize the externality, however, with the payment forming part of GNI. Pagiola (2008) provides an example for Costa Rica.

necessary data can be found, measuring the insurance value of biodiversity would in fact increase the estimated total wealth of the economy. Note, however, that this would not increase measured GNI in the national accounts, although it could be argued that it increases *expected* GNI in the face of some known probability of a catastrophe of a given size.

Placing existence (or bequest) values on the biodiversity assets of a country would immediately take you outside the SNA boundary.[7] From an accounting perspective this would increase measured consumption to include the dollar value of the flow of benefits tied to the existence of species, while the balance sheet would expand to include the capitalized value of this flow. As long as what is measured is 'pure' existence value (i.e. completely excluding any of the other values just discussed), there would be no double-counting in the accounts. Existence values are obviously another form of externality, since the households benefiting from the existence of the biodiversity are not paying a user fee to the owner of the asset.

As this discussion makes clear, biodiverse natural areas provide multiple flows of benefits. The capitalized values of these benefits can in principle be added up without double-counting in the balance sheet.

This discussion has concentrated on the diverse values of natural areas that could be valued in the balance sheet, focusing on those values that are linked to biodiversity. It has not talked specifically about the value of biodiversity in the balance sheet, and this is because of the difficulty in decomposing the value of natural areas into their different sources of value. While bioprospecting and knowledge values may be specific to the quantity and character of biodiversity, this is less obviously true for natural areas as a source of user fees, ecotourism rents, and ecosystem services—what is being valued is a composite good made up of diverse components including soil, hydrology, geology, topography, climate, location, and biodiversity, as noted earlier.

It may be possible, however, to derive many of the *marginal* values of the specific properties of a natural area. This is discussed in the next section.

9.5 MEASURING NET INCOME AND NET SAVING

The link between the wealth inherent in natural areas and GNI is a natural starting point for thinking about biodiversity, net income, and net saving. A key point is that many of the economic benefits derived from biodiverse

[7] Since stated-preference methods are the tools required to establish existence values, measuring the total existence value of the suite of biodiversity possessed by a country would also be subject to well-known difficulties, including the evidence that survey respondents have difficulty putting rational values on individual species vs. collections of species.

natural areas are already captured in GNI—user fees, ecotourism expenditures, bioprospecting rents, the economic value of knowledge derived from biodiversity, and the external benefits provided by ecosystem services are all included in GNI. As just noted, the insurance services provided by biodiversity are probably not reflected in GNI; nor are the flows of existence values since they lie outside the SNA boundary.

From an analytical perspective, one of the principal reasons for building the national balance sheet account is to shed light on the sustainability of development. Pearce and Atkinson (1993) were the first to present empirical estimates of net national saving adjusted for resource depletion and damage to the environment. They interpret the adjusted saving measure as an indicator of sustainable development, and subsequent growth-theoretic papers by Hamilton and Clemens (1999), Dasgupta and Mäler (2000), and Asheim and Weitzman (2001) provided the theoretical underpinning—net (or 'genuine') saving, calculated as the net change in real wealth, is a dollar-valued measure of the change in social welfare. Negative genuine saving indicates that the country is on an unsustainable path.

An overarching issue in measuring genuine saving is the need to measure the change in *real* asset values, measured as a fixed price times the quantity which is added to or subtracted from the underlying stock. The relevant price is the shadow price of a unit of the asset, or the change in social welfare associated with a marginal change in the quantity of the asset.

Barbier (2012) provides a useful overview of how production function techniques can be used to measure marginal values of ecosystem services, and we generalize this to the case of biodiversity in Appendix I. The basic idea is that we can conceive the flow of benefits provided by a natural area as the result of a 'natural production function', where the inputs to production are the different properties of the natural area, including biodiversity as measured (for example) by relative species abundance. Changes in the flow of benefits as well as the associated changes in the properties of the ecosystem can be used to model the production function econometrically, and the coefficients on the inputs to the production function represent the marginal values of the different ecosystem properties.

While it is simple to write out the maths for this technique, as shown in Appendix I, Barbier (2007) enumerates the complexities involved. In particular, this assumes the availability of data on changes in the economic benefits derived from a natural area as well as the associated changes in the components and properties of the natural area, including, in this case, the quantity of biodiversity. A quantitative understanding of the structure and function of the ecosystem is required. Market imperfections have to be taken into account, as well as stock effects when changes in ecosystem services are sufficiently large. For insurance services provided by biodiversity, the production function approach requires the estimation of the damages to human well-being avoided by having biodiverse natural areas.

If the SNA boundary were extended and existence values were brought into the national balance sheet, many of the difficulties in measuring existence value using stated-preference methods would come to the fore. And it is unclear whether the total existence value would fall if a single species became (locally) extinct—for example, people might value the existence of other species more as a result of the extinction. Existence values may simply not lend themselves to marginal valuation.

Conversely, if values of the benefits from the use of natural areas, based on surveys of willingness to pay, were brought into imputed measures of total wealth and national income, it would be possible to value changes in natural areas (including biodiversity) as long as a key proviso noted in the previous section applies—it must be the case that stated willingness to pay varies systematically with changes in the properties of the natural area. If this were the case, then in principle the relevant portion of the utility function for the users of natural areas could be estimated, yielding marginal values of the different properties of the natural area.

While we concluded the previous section by noting the likely difficulties in arriving at any additive decomposition of the values of natural areas into their component properties, including biodiversity, production function techniques offer a way to measure marginal changes in the flow of benefits from natural areas associated with marginal changes in the properties of the natural area. In principle this opens the door to adjusting the measure of genuine saving to reflect losses of biodiversity, but the practical difficulties should not be underestimated. These difficulties in measuring genuine saving spill over to measuring net income, which is defined as the sum of consumption and net (genuine) saving.

We now turn to an analysis of the World Bank's estimates of the value of protected areas.

9.6 VALUING CONSERVATION IN NATIONAL ACCOUNTS: EMPIRICAL ESTIMATES

As the foregoing has emphasized, biodiversity can provide a range of functions of value to human well-being. In theory and in practice this contribution to well-being can be valued, but what is lacking are comprehensive estimates of these values both within and across countries—the literature on valuing biodiversity is generally concerned with specific ecosystems in specific countries.

As a proxy for biodiversity values, we analyse the values of terrestrial protected areas across countries, exploiting figures published in *The Changing Wealth of Nations* (World Bank, 2011). This is clearly not the same as valuing biodiversity, but there is at least a link: protected areas harbour much, but certainly not all, of the world's biodiversity. However, just as there are no comprehensive measures of biodiversity values across countries, there is a similar lack of cross-country estimates of the value of natural areas based on the value of the environmental services provided by these areas. The World Bank therefore falls back on quasi-opportunity costs as the basis of valuation.

Starting from estimates of the extent of terrestrial protected areas from the World Conservation Monitoring Centre (as published in the *World Development Indicators*), World Bank (2011) assumes that the best alternative use for protected land is in agriculture. It further assumes that areas subject to protection are not the most productive agricultural lands—first, because most such conversions to agriculture would already have occurred over the course of history; and second, because it is unlikely that a government would choose to declare prime agricultural land as a protected reserve. World Bank (2011) therefore values protected areas at the same per-hectare value as what is, on average, the least productive agricultural land in a given country—hence the notion of a 'quasi-opportunity cost'.[8] The assumption is that the government establishing a protected area values the newly protected land at least as much as if it were in its alternative agriculture use.

What we analyse in the following sub-sections is therefore easily a third-best measure of the value of biodiversity in national balance sheets. But the analysis is not without interest, and it leads to some tentative policy conclusions.

9.6.1 Protected area rents in relation to GDP

While World Bank (2011) publishes the asset value of protected areas—it forms part of the aggregate value of natural capital and, more broadly, comprehensive wealth—here we focus on the annual land rent from protected areas ('PA rents') per capita.[9] The natural question to ask is how this is related to GDP per capita across countries, and Figure 9.1 provides the answer.

[8] In practice, World Bank (2011) uses the lower of cropland and pastureland average per-hectare values in each country. For simplicity we refer to 'opportunity costs' in lieu of 'quasi-opportunity costs' in what follows.

[9] World Bank (2011) attempts to value the wealth available to the current generation, and therefore calculates land asset values as the present value of land rents over a 25-year horizon.

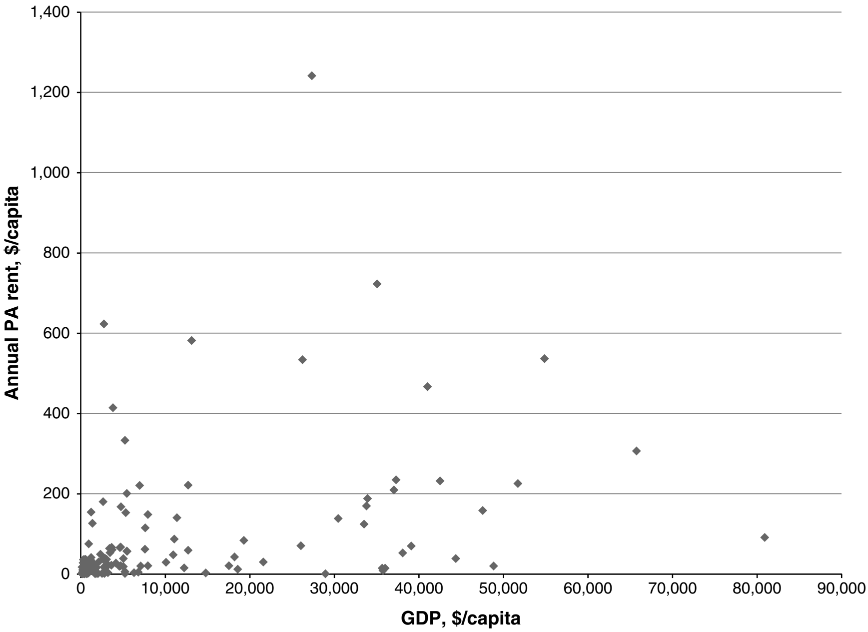

Fig. 9.1. PA rents per capita vs. GDP per capita, 2005

Source: Derived from data published in World Bank (2011).

The scatter in Figure 9.1 shows no discernible trend or pattern, and this is not entirely surprising. While land rents per hectare will tend to rise with GDP per capita, owing to better technology, there is no obvious reason why the underlying endowment (land suited to be protected areas) should vary with income. There is conceivably a greater willingness of high-income countries to establish protected areas than in developing countries, but we show in what follows that this is not borne out by the data.

The next obvious question to examine is the variation in PA rents as a percentage of GDP across countries.

9.6.2 PA rents as a percentage of GDP

Figure 9.2 shows the distribution of the PA rent percentage across the 146 developed and developing countries in the sample from World Bank (2011).

The distribution is highly skewed, with PA rents comprising 1 per cent or less of GDP in 55 per cent of the countries in the sample, and 2 per cent or less of GDP in 76 per cent of the countries in the sample. With some exceptions, therefore, PA rents tend to be a small share of GDP. This leads to the next

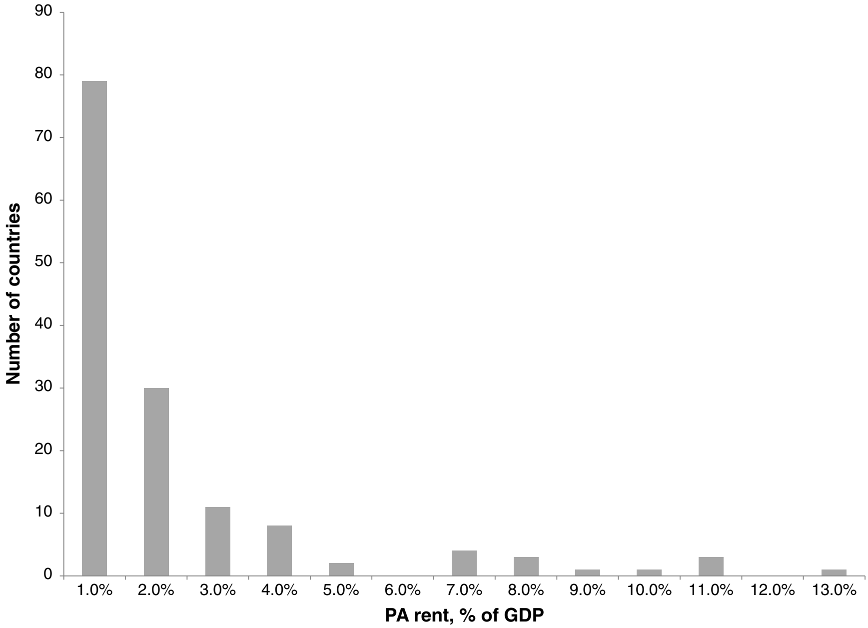

Fig. 9.2. Distribution of PA rents as a percentage of GDP, 2005

Source: Derived from World Bank (2011). Note that Ecuador (22.6%) is not shown.

analytical question: how do these shares of GDP vary across income classes of countries?[10]

9.6.3 PA rents as a share of GDP across income classes

Appendix II Tables A9.1–A9.3 present the detailed figures on GDP per capita, PA asset values per capita, and PA rents as a share of GDP for each of the countries in the sample, classified by income category. Using 1 per cent and 5 per cent of GDP as thresholds, Table 9.1 summarizes the findings.

The striking result in Table 9.1 is the high proportion of developing countries where PA rents exceed 1 per cent of GDP, often by a large amount. Only four OECD countries—Sweden (1.1 per cent), Slovak Republic (1.2 per cent), Canada (2.1 per cent), and New Zealand (4.5 per cent)—had shares of GDP above 1 per cent.

[10] We use the standard income classes of the World Bank, low and middle income, as well as OECD countries.

Table 9.1. Countries with PA rents > 1% of GDP, by income class, 2005

	Low income	Middle income	OECD
# of countries in class	41	65	29
# of countries in class with PA rents > 1% of GDP	24	34	4
% of countries in class with PA rents > 1% of GDP	59%	52%	14%
Which countries had PA rents > 5% of GDP?	Benin (6%) Kenya (7%) Lao PDR (7%) Tajikistan (8%) Nepal (9%) Ethiopia (10%) Uganda (11%)	Dominica (6%) Thailand (7%) Cameroon (8%) Honduras (9%) Belize (11%) Bhutan (12%) Ecuador (23%)	

Source: Author's calculations based on World Bank (2011).

9.6.4 Explaining the high PA rents as a share of GDP in developing countries

Table 9.2 lays out three potentially relevant indicators to explain this high share of PA rents in the GDP of developing countries. The first two are physical measures—hectares of protected area per capita and protected area as a percentage of total land area. The third is a measure derived from the comprehensive wealth accounts published in World Bank (2011)—land values as a share of total wealth.

The physical indicators are clearly not the explanatory variables we are seeking—protected area per capita in developing countries is about two-thirds of the value in OECD countries, while there is little variation in the protected area share of total land area across the different income classes. The third indicator tells the story. Land values (excluding urban land) make up only 1.4 per cent of the total wealth of OECD countries, while it is ten to twenty times higher as a proportion of wealth in middle- and low-income countries. Since the opportunity cost of creating a protected area is based on the rents from agricultural land, this will necessarily be a much larger share of GDP in developing countries compared with OECD countries.

9.6.5 Policy implications and some caveats

Protected areas and the biodiversity they harbour provide both local and global benefits. The institution that finances environmental global public goods is the Global Environment Facility (GEF), and it is worth assessing the level of finance that it disburses for biodiversity protection.

Table 9.2. Protected area extent and land values by income class, 2005

	Low income	Middle income	OECD
Protected area per capita, ha	0.22	0.21	0.34
Protected area as a percentage of total land area	10.8%	10.6%	11.8%
Land value (cropland, pastureland, forested land, protected areas) as a percentage of total wealth	29.5%	12.8%	1.4%

Source: Derived from World Bank (2011) and *World Development Indicators*.

According to the GEF's *Annual Report 2010*, the total finance it provided for biodiversity conservation from 1991 to 2010 amounted to $3.1 billion, while this amount leveraged a further $8.4 billion in co-finance from development partners. In total, therefore, GEF finance and co-finance averaged $575 million per year for biodiversity conservation over these two decades. This finance is provided to both low- and middle-income countries, but the bulk of it is allocated to the mega-diverse middle-income giants: Brazil, China, and Indonesia.

We can put this figure in context by measuring the PA rents for *low-income* countries as a whole in 2005—this is the total annual opportunity cost of conservation in these countries. This amounts to $7.6 billion. The average annual GEF finance plus co-finance to both *low- and middle-income* countries is therefore only 8 per cent of this figure.

Before we jump to the conclusion that biodiversity conservation in low-income countries is woefully under-financed, some caveats are in order. First, the GEF role is to finance global environmental public goods, not the local benefits that countries derive from their conservation efforts. So the GEF financing figure *should* be lower than the local opportunity cost of conservation.

Second, World Bank (2011) bases all of its valuation of natural resources on the basis of world prices, as a way to level the playing field when country comparisons are being made. Of course, this is also the appropriate shadow price to use when deriving the value of natural assets. However, protected areas, by their very nature, tend to be far from markets and infrastructure in developing countries. As a result, the opportunity costs of conservation derived in World Bank (2011) probably exceed the opportunity cost of conservation at local prices in these countries.

Third, the assumption that protected areas are only as productive as the least valuable category of agricultural land in a given country (the quasi-opportunity cost) may be too strong. If the fact that a given natural area has not been developed for agriculture to date reflects revealed preference—local people know that its agricultural productivity would be very low—then the opportunity costs used in World Bank (2011) would be correspondingly too high.

Granting these caveats, the 8 per cent ratio of GEF biodiversity finance plus co-finance to the World Bank's estimate of low-income country opportunity costs for conservation is extremely low, particularly since the bulk of this finance is going to middle-income countries.

9.7 CONCLUSIONS ON BIODIVERSITY AND NATIONAL ACCOUNTING

Early in this chapter we quoted the SNA definitions pertaining to the value of natural resources in the balance sheet account for a country. The basic idea is that the natural resource must produce an economic benefit to its owner in order to be considered an asset. While natural (undeveloped) areas are generally owned by governments, countries building balance sheet accounts have tended to exclude the value of government land in the balance sheet accounts. We present examples of how biodiverse natural areas can provide economic benefits to governments, and therefore could be valued in the balance sheet account.

The literature on the economics of biodiversity points to a range of sources of economic value linked to some quantitative measure of biodiversity. Some of these values, particularly bioprospecting and knowledge values, may be directly linked to specific 'biodiversity assets', and so could be measured directly in the national balance sheet. For natural areas as a source of user fees, ecotourism rents, and ecosystem services, it is much less obvious that the natural area as a composite good lends itself to a decomposition of its total value into specific values associated with the different properties of the natural area, including biodiversity. A specific 'value of biodiversity' in the national balance sheet may therefore be an elusive goal, but an aggregate value of natural areas may be feasible.

On the other hand, given sufficient knowledge of natural systems and sufficient variation in the data on the inputs and outputs of natural systems, it may be possible to derive production functions for the different categories of value provided by natural areas. This in turn could lead to values of the marginal benefits associated with changes in the quantity of biodiversity, and therefore to adjusted measures of net income and net (genuine) saving in an extended national accounting system.[11]

A number of complicating factors arise when it comes to including the value of natural areas in national balance sheet accounts. One important issue is that

[11] This ability to measure the marginal values of properties such as biodiversity suggests that constant returns to scale in the production function could lead to an additive decomposition of all the sources of value which could be broken out in the balance sheet account. However, constant returns to scale seem unlikely given the highly non-linear character of many natural systems. This non-linearity also implies that the estimated marginal values of different properties of a natural system are purely local.

many user fees are not capturing the full willingness to pay of the user of the natural area. Under fairly stringent conditions—construction of a demand curve for person-days of use of the natural area, where the price is sensitive to any changes in the properties of the natural area—it would be possible to impute augmented values of national wealth and national income based on willingness to pay to use natural areas.

The other big issue is that many natural areas provide benefits as externalities to the wider economy. The consequence of these externalities for wealth measurement is that the inclusion of many ecosystem services in the balance sheet would lead to double-counting. The role of valuing ecosystem services in national accounting is therefore typically to decompose the sources of value underpinning those assets (such as farmland) which benefit from external environmental services.

This external nature of many ecosystem services has policy consequences. The natural areas which are sources of ecosystem services will generally be at risk, since the owner of the natural area (a government or a private actor) may not be aware of, or may not care about, its value to other actors when development decisions are being made. PES schemes are a policy response which internalizes the externality.

On quantification, the analysis of the (quasi-)opportunity costs of conserving natural areas based on wealth accounting data in *The Changing Wealth of Nations* (World Bank, 2011) sheds light on the disproportionate burden of conservation costs as a share of GDP being borne in developing countries in comparison with OECD countries. The reason for this is the much larger share of total wealth (and therefore national income) that is derived from land in developing countries compared with OECD countries.

To put these values of protected areas in context, the annual GEF finance and co-finance for biodiversity conservation in all developing countries was, based on a 20-year average, only 8 per cent of the opportunity cost of conservation in low-income countries in 2005. This indicates that the opportunity cost of conservation borne by low-income countries is not being substantially offset by international flows of concessional finance.

On balance, there is considerable scope to include the use values of natural areas in national accounts. This is because there are flows of user fees and rents associated with the use of natural areas. As a property of natural areas, biodiversity can be implicitly or explicitly valued in the balance sheet accounts, depending on the nature of the benefits it provides (e.g. use values for nature tourists vs. bioprospecting fees), while marginal changes in biodiversity may be valued depending on the availability of sufficient data to estimate the associated changes in the production of benefits which people value. Non-use values appear to be much less amenable to valuation in the national accounts, owing to measurement issues and the inherently non-marginal nature of some of these values.

APPENDIX I: DERIVING MARGINAL VALUES OF BIODIVERSITY USING PRODUCTION FUNCTION TECHNIQUES

Suppose that a given natural area generates a total annual flow of dollar-valued benefits W, and that this flow is an increasing function of a set of properties of the natural area, N_i, and biodiversity measured, for example, as relative species abundance. $w(N_i, B)$ can then be conceived as a production function for the value of the natural area. If there is a decrease in the quantity of biodiversity ΔB, the change in the real value of the natural area is given by:

$$-\frac{\partial w}{\partial B}\cdot\Delta B \tag{9.1}$$

The partial derivative, which can in principle be estimated econometrically, represents the marginal value of a unit of biodiversity. As a marginal value, this can be used in national accounting, in particular in an adjusted measure of net saving in the economy.

If the natural area is providing a flow of benefits as an externality to another productive activity, such as water regulation services to farmland, for example, then the appropriate model is a nested production function. Assume the value of crop production from farmland is given by $F(N_i, w)$, where now the N_i are both the properties of the farmland as well as inputs of capital, labour, and intermediate goods such as fertilizer. $w = w(N_j, B)$ is the production function for water regulation services, which depend on the properties of the ecosystem N_j and its biodiversity B. If there is a decline in the biodiversity of the natural area ΔB, the change in the value of farm production associated with this decline is therefore:

$$-\frac{\partial F}{\partial w}\cdot\frac{\partial w}{\partial B}\cdot\Delta B \tag{9.2}$$

APPENDIX II: DATA ON PROTECTED AREA ASSET VALUES PER CAPITA AND LAND RENTS

Table A9.1. Selected low-income countries, 2005 (US$)

	GDP/cap	PA asset value/cap	PA rent % GDP
Bangladesh	429	16	0.24%
Benin	562	562	6.40%
Burkina Faso	385	211	3.51%
Burundi	154	13	0.53%
Chad	542	140	1.66%
Comoros	602	330	3.51%
Congo, Dem. Rep.	125	19	0.96%
Ethiopia	165	261	10.13%
Gambia, The	415	10	0.16%

(*continued*)

Table A9.1. continued

	GDP/cap	PA asset value/cap	PA rent % GDP
Ghana	496	18	0.23%
Guinea	325	27	0.53%
Guinea-Bissau	419	85	1.29%
Haiti	444	4	0.05%
India	732	145	1.27%
Kenya	526	557	6.78%
Kyrgyz Republic	476	96	1.28%
Lao PDR	475	554	7.47%
Lesotho	662	1	0.01%
Liberia	170	16	0.60%
Madagascar	282	41	0.94%
Malawi	215	60	1.78%
Mali	403	64	1.02%
Mauritania	717	89	0.80%
Moldova	831	56	0.43%
Mozambique	317	12	0.25%
Nepal	298	433	9.30%
Niger	262	160	3.91%
Nigeria	803	17	0.13%
Pakistan	691	286	2.65%
Papua New Guinea	804	319	2.54%
Rwanda	281	114	2.59%
Senegal	800	102	0.82%
Sierra Leone	240	8	0.20%
Sudan	691	295	2.73%
Tajikistan	358	434	7.75%
Togo	391	39	0.64%
Uganda	325	558	10.98%
Uzbekistan	547	101	1.18%
Vietnam	642	152	1.52%
Zambia	626	100	1.02%
Zimbabwe	458	79	1.10%

Source: Derived from World Bank (2011).

Table A9.2. Selected middle-income countries, 2005 (US$)

	GDP/cap	PA asset value/cap	PA rent % GDP
Albania	2,666	574	1.38%
Algeria	3,112	384	0.79%
Angola	1,857	80	0.27%
Argentina	4,736	320	0.43%
Armenia	1,598	373	1.49%
Azerbaijan	1,578	212	0.86%

Belarus	3,090	560	1.16%
Belize	3,821	6,468	10.84%
Bhutan	1,242	2,407	12.40%
Bolivia	1,044	443	2.72%
Botswana	5,468	888	1.04%
Brazil	4,743	1,042	1.41%
Bulgaria	3,733	938	1.61%
Cameroon	945	1,165	7.89%
Cape Verde	2,055	17	0.05%
Chile	7,631	1,793	1.50%
China	1,731	107	0.40%
Colombia	3,404	993	1.87%
Congo, Rep.	1,723	10	0.04%
Costa Rica	4,633	1,026	1.42%
Côte d'Ivoire	908	39	0.28%
Croatia	10,090	445	0.28%
Dominica	5,247	5,206	6.35%
Dominican Republic	3,670	1,028	1.79%
Ecuador	2,751	9,723	22.62%
El Salvador	2,825	18	0.04%
Fiji	3,655	333	0.58%
Gabon	6,322	49	0.05%
Georgia	1,470	242	1.06%
Grenada	6,818	66	0.06%
Guatemala	2,140	463	1.38%
Guyana	1,105	160	0.93%
Honduras	1,412	1,965	8.91%
Indonesia	1,258	411	2.09%
Iran, Islamic Rep.	2,754	267	0.62%
Jamaica	4,179	426	0.65%
Jordan	2,326	759	2.09%
Latvia	6,973	3,444	3.16%
Lithuania	7,604	958	0.81%
Macedonia, FYR	2,937	235	0.51%
Malaysia	5,499	879	1.02%
Mauritius	5,054	288	0.36%
Mexico	7,973	316	0.25%
Mongolia	991	443	2.86%
Morocco	1,931	18	0.06%
Namibia	3,491	826	1.52%
Nicaragua	1,166	549	3.01%
Panama	4,776	2,611	3.50%
Peru	2,881	603	1.34%
Philippines	1,205	302	1.60%
Poland	7,963	2,306	1.85%
Romania	4,572	297	0.42%
Russian Federation	5,337	2,380	2.85%
South Africa	5,234	93	0.11%
Sri Lanka	1,242	640	3.30%
St Vincent and the Grenadines	5,070	599	0.76%
Swaziland	2,540	17	0.04%

(*continued*)

Table A9.2. continued

	GDP/cap	PA asset value/cap	PA rent % GDP
Syrian Arab Republic	1,561	63	0.26%
Thailand	2,644	2,813	6.81%
Tunisia	3,219	51	0.10%
Turkey	7,088	310	0.28%
Ukraine	1,829	266	0.93%
Uruguay	5,252	19	0.02%
Vanuatu	1,862	251	0.86%
Venezuela, RB	5,475	3,136	3.67%

Source: Derived from World Bank (2011).

Table A9.3. Selected OECD countries, 2005 (US$)

	GDP/cap	PA asset value/cap	PA rent % GDP
Australia	33,945	2,932	0.55%
Austria	37,067	3,272	0.57%
Belgium	36,011	222	0.04%
Canada	35,088	11,293	2.06%
Czech Republic	12,706	924	0.47%
Denmark	47,547	2,463	0.33%
Finland	37,319	3,659	0.63%
France	33,819	2,646	0.50%
Germany	33,543	1,935	0.37%
Greece	21,621	458	0.14%
Hungary	10,937	740	0.43%
Iceland	54,885	8,382	0.98%
Ireland	48,866	304	0.04%
Israel	19,330	1,300	0.43%
Italy	30,479	2,158	0.45%
Japan	35,781	128	0.02%
Korea, Rep.	17,551	322	0.12%
Luxembourg	80,925	1,413	0.11%
Mexico	7,973	316	0.25%
Netherlands	39,122	1,082	0.18%
New Zealand	27,354	19,395	4.54%
Norway	65,767	4,788	0.47%
Portugal	18,186	655	0.23%
Slovak Republic	11,385	2,190	1.23%
Spain	26,056	1,095	0.27%
Sweden	41,041	7,284	1.14%
Switzerland	51,734	3,521	0.44%
United Kingdom	38,122	815	0.14%
United States	42,516	3,625	0.55%

Source: Derived from World Bank (2011).

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Part IV

International and Development Aspects

10

Biodiversity, Poverty, and Development: A Review

Charles Palmer and Salvatore Di Falco

10.1 INTRODUCTION

The Millennium Ecosystem Assessment (MA, 2005) was a landmark attempt to assess the state of the world's ecosystems and the consequences of ecosystem change for human well-being. It found that the structure and functioning of global ecosystems have changed more rapidly between 1950 and 2000 than at any comparable period in human history. During this fifty-year period, the world's population doubled while the global economy grew sixfold, leading to rapid increases in demand for ecosystem goods and services. For example, food production more than doubled while wood harvests for pulp and paper production tripled. These increases in demand were met by increasing the supply of ecosystem goods and services.

Biodiversity is crucial for the production of a range of ecosystem goods and services, from timber, meat, and medicines to hydrological (water) services, soil management, and biosphere resilience. A growing body of evidence shows how it supports system productivity and how its loss can negatively affect ecosystem functioning (e.g. Loreau and Hector, 2001; Naeem et al., 1994; Cork et al., 2002; Hooper et al., 2005; Tilman and Downing, 1994; Zhu et al., 2000; Landis et al., 2008). Entering into force in 1993, the Convention on Biological Diversity (CBD) declared the conservation of biological diversity 'a common concern of humankind' and an integral part of economic development.

In this chapter, we examine the evidence for the role of land-based biodiversity and biodiversity conservation in economic development and poverty, at both the macro (e.g. country) and micro level (e.g. farm). Biodiversity plays an important role not just in social systems but also ecological ones.

Examination of the evidence at a smaller scale enables a more precise exploration of *how* biodiversity influences ecosystem services that are ultimately instrumental for development—e.g. in supporting food security. Our focus is on those areas and countries with high endowments of biodiversity, which also tend to be located in poorer or 'less-developed' countries (Fisher and Christopher, 2007; Barrett et al., 2011).

We begin, in section 10.2, with background to the themes covered in the chapter, along with the definitions used. In section 10.3, we present evidence for general relationships between biodiversity and economic development before providing a closer examination of the links between biodiversity and ecosystem services. Key for food production, *agricultural* biodiversity is used to illustrate these links. Given projected, future population increases and continued economic growth and consumption, section 10.4 discusses research on future scenarios for biodiversity and development. Future threats to biodiversity need to be addressed via effective policies and strategies to protect biodiversity. Section 10.5 first examines evidence of a relationship between biodiversity protection and economic growth at the country scale, before focusing on the very low incomes of rural people in biodiverse developing countries. We then present evidence for two types of policy—protected areas and bioprospecting—which aim to protect biodiversity, and their impacts on the rural poor. Section 10.6 concludes.

10.2 BACKGROUND AND DEFINITIONS

For many countries, ecosystem goods and services have long contributed to human well-being and economic development—the latter defined in terms of the growth of GDP. Agriculture, fisheries, and forestry have long been crucial to countries' development strategies, providing capital for investments in other sectors and for the alleviation of poverty (MA, 2005). Consumption goods such as timber, fuel, meat, and medicines are ones that are typically included in countries' national accounts. Agriculture in 2000, for example, provided work and income-earning opportunities for half the world's total labour force, and accounted for almost a quarter of GDP in countries with incomes of less than US$765 per capita.

The production of ecosystem goods and services that exploit biodiversity for direct human consumption has the potential to be sustainable but instead they are often over-harvested and degraded (Albers and Ferraro, 2006). Over-harvesting, along with land-use change, climate change, invasive species, and pollution have all contributed to biodiversity loss, in particular, the conversion of large areas of natural ecosystems to agriculture (MA, 2005). Biodiversity loss is such that the Earth could be in the midst of its sixth 'mass extinction'

(Barnosky et al., 2011).[1] The MA (2005) documents this loss in great detail, for instance, in terms of declines in populations and indeed whole populations of known species. The global extinction rate may have risen by as much as 1,000 times over background rates typical during the Earth's history.

'Biodiversity' encompasses a range of levels, scales, and attributes, and is thus impossible to capture in a single measure. In general, it defines the variety of living things in terms of genes (the smallest unit) within species, species themselves, and ecosystems (the largest unit). The latter refers to the typically complex interrelationships between species and their habitats.[2] Due to this complexity, biodiversity policies tend to focus on simple indicators such as the amount of land under protection or the International Union for Conservation of Nature (IUCN) Red List indicators (lists of vulnerable species). Only a proportion of all species has been taxonomically classified. Estimates of total number of species abound. However, these range so widely that it is clear that no one really knows the true global number of species (see Wilson, 1986). As a result, number of species is considered neither a reliable nor a particularly useful measure for informing biodiversity policy (Albers and Ferraro, 2006).

However biodiversity is defined, it plays an important role in the production of a wide range of ecosystem services, which do not have market prices and are excluded from GDP data (see Chapter 6 of this book by Atkinson et al.). Yet such services are critical to human economies and societies, for example, ensuring that ecosystems can withstand changes, particularly shocks, from both natural and human systems. People also value certain species simply for existing. Biodiversity may also have some future commercial value—e.g. new pharmaceutical products derived from the discovery of a yet-to-be discovered species of plant. While all these benefits have value they do not have a market price, which implies an important role for policies that invest in and protect biodiversity. In recent decades, a range of policy initiatives and instruments, implemented at different scales (international, national, sub-national) by international agencies, national governments, NGOs, and so on, have been implemented with the aim of protecting biodiversity. These include 'bioprospecting' and protected area networks, such as national parks. Bioprospecting involves searching for, collecting, and deriving genetic material from wild species that can be used in commercialized pharmaceutical, agricultural, industrial, or chemical processing end-products.

[1] Of the four billion species estimated to have evolved on Earth over the last 3.5 billion years, some 99 per cent no longer exist. Thus, extinction is relatively common, although it is balanced by the emergence of new species (Barnosky et al., 2011). Mass extinction occurs when extinction rates accelerate relative to origination rates such that over 75 per cent of species disappear within a geologically short interval—typically less than two million years, in some cases much less (Barnosky et al., 2011).

[2] Ecosystem complexity can be characterized by, for example, the possibility of sudden, unpredictable non-linear changes, feedback effects, and vulnerability to sudden shocks such as from fire or disease (Barrett et al., 2011).

Such policies have to contend with the fact that biodiversity is unevenly distributed across the world. In particular, much biodiversity is concentrated in less-developed countries.[3] For example, around half of all land-based species are located in one-tenth of the Earth's land surface, with many found in areas of tropical forests (Wilson, 1986). Biodiversity 'hot spots' tend to be concentrated in rural areas where people's livelihoods depend disproportionately on the exploitation of forests, rangelands, soils, water, and wildlife (Myers et al., 2000; Barrett et al., 2011). The challenge of policy is thus not only to protect biodiversity but also to ensure that people are not made worse off as a result of policy implementation.

In summary, biodiversity clearly contributes to economic growth and development, with evidence of a trade-off between these. The next section, 10.3, first examines the research undertaken on the relationship between growth and biodiversity before looking at the more complex relationships between biodiversity and ecosystem services. Given recorded losses in biodiversity in recent decades, and the multiple drivers of these losses, section 10.4 reviews research that generates projections of future losses and their drivers. In general, these have widely varying predictions. Of course, such analyses are based on numerous assumptions. One relates to the prevailing policy environment in which biodiversity is addressed. Section 10.5 reviews two important policies for biodiversity conservation, one driven by the public sector (protected areas) and one by the private sector (bioprospecting). Bioprospecting has not been adopted as widely as its proponents have predicted. Protected area networks, on the other hand, continue to expand and are the commonest means of biodiversity conservation around the world. However, there are concerns regarding their relative effectiveness and their impacts on the rural poor.

10.3 BIODIVERSITY, ECOSYSTEM SERVICES, AND DEVELOPMENT

In this section, we begin with an examination of the evidence for a direct relationship between biodiversity and conventional economic growth and

[3] In 1998, the NGO Conservation International identified seventeen countries with exceptional endowments of biodiversity. Excluding Australia and the USA, the political group Like-Minded Megadiverse Countries (LMMC) brought together the interests and concerns of fifteen of these countries plus other less-developed countries, in 2002. It claims to represent about 80 per cent of the world's biodiversity, and 45 per cent of the world's population. Members include: Bolivia, Brazil, China, Colombia, Costa Rica, Democratic Republic of the Congo, Ecuador, India, Indonesia, Kenya, Madagascar, Malaysia, Mexico, Papua New Guinea, Peru, Philippines, South Africa, and Venezuela (Deke, 2008). While acknowledging differences in per-capita incomes among these countries, these are important examples of biodiverse 'less-developed' or 'developing' countries, as defined in this chapter.

development. Yet, understanding the relationship between biodiversity and incomes requires knowledge of *how* biodiversity influences the production of ecosystem services since the latter are ultimately instrumental for development. On their own, relationships between measures of biodiversity and incomes may be insufficient for drawing meaningful policy guidance. The relationship between biodiversity and many different types of ecosystem service is complex and hence, not well understood (Albers and Ferraro, 2006). One crucial exception is the relationship between biodiversity and agricultural production. We review an emerging body of evidence that sheds light on this relationship.

10.3.1 Biodiversity, and economic growth and incomes

The relationship between per-capita income and measures of environmental degradation has come to be framed according to the Environmental Kuznets Curve (EKC) hypothesis (Dasgupta et al., 2002). This follows an inverted U-shaped relationship in which environmental degradation initially rises with increasing incomes but then, at some level of income, subsequently declines.[4] At higher income levels, a shift to less environmentally degrading economic activities, more effective environmental regulation, and a shift in social attitudes to reduce environmental degradation begins to drive improvements in environmental quality. Thus, a trade-off between economic growth and environmental quality is hypothesized to exist only when countries are relatively poor. There is, however, limited empirical evidence in support of the EKC. This evidence tends to focus on various measures of air pollution (e.g. Selden and Song, 1994; Grossman and Krueger, 1995). A limited number of studies have explored the relationship between economic growth and biodiversity loss.

Naidoo and Adamowicz (2001) examine the link between numbers of threatened species, as classified by the IUCN, and per-capita GNP, using data from over 100 countries. Their main finding is that increasing GNP appears to be associated with reductions in threatened species numbers for some taxa, particularly birds. Yet, for most of the other groups surveyed, including plants, mammals, amphibians, reptiles, fish, and invertebrates, they find no evidence for a relationship between numbers of threatened species and GNP.

[4] The EKC is based on analogy with the observations of Kuznets (1955), who explored the U-shaped relationship between income inequality and changes in per-capita income. Hepburn and Bowen (2012) provide a recent conceptual and synthetic analysis of the relationship between economic growth and environmental limits, including more discussion on the conceptual and empirical relevance of the EKC.

Note, however, that Naidoo and Adamowicz (2001) focused on the relationship between per-capita income and the relative degree of extinction threat. Hence, they did not measure biodiversity losses per se. Dietz and Adger (2003) investigate the relationship between economic growth, biodiversity loss, and efforts to conserve biodiversity using a combination of data for a sample of countries containing biodiverse tropical forest. More in keeping with the EKC hypothesis, they suggest that if economic growth drives biodiversity losses, e.g. through destruction of habitats, then the data should reveal that as economies grow biodiversity losses intensify. But where increasing incomes are associated with an increase in demand for biodiversity conservation then investments in biodiversity protection should rise, leading to a corresponding decline in biodiversity loss.

Dietz and Adger, however, fail to find an EKC between income and rates of species loss. Moreover, they do not find an EKC between income and rates of habitat loss. Mills and Waite (2009) reanalysed the data used by Dietz and Adger and attempted to address some of the statistical issues associated with the dataset. Despite finding some initial support for the EKC, the overall conclusion is that there is relatively little empirical evidence of an EKC between income and biodiversity.

More recently, Perrings and Halkos (2010) examine the relationship between gross national income (GNI) per capita and threats to biodiversity. Using the number of species in each taxonomic group under threat (according to the 2004 IUCN Red List), they model the relationship between the level of threat and GNI per capita in a sample of seventy-three countries. They find an empirical relationship, which provides support for the EKC hypothesis for four groups of species: mammals, birds, plants, and reptiles. Income growth is found to be strongly correlated with increasing levels of threat to biodiversity. This may be due to the dependence of poorer countries' economies on agriculture. Thus, income growth depends on the expansion of agricultural lands into more 'marginal' areas, which are often habitats for wild species as well. It should be stressed, however, that the nature of this analysis raises some concerns about the statistical robustness of these results, and how they might be interpreted.

10.3.2 Biodiversity and ecosystem services

Agricultural biodiversity refers to all diversity within and among species found in crop and domesticated livestock systems (Qualset et al., 1995; Wood and Lenné, 1999). Similar to species hotspots, 'centres of origin/diversity', i.e. hotspots of wild genetic diversity, have been identified for major crop plants, which also tend to be concentrated in tropical and sub-tropical regions. Domesticated biodiversity (i.e. crops) is located in agricultural landscapes (*in situ*). It is complemented by wild relatives stored in gene banks and breeders' collections (Smale, 2006). Biodiversity is utilized by farmers and breeders to adapt crops to different and changing production environments. Crop biodiversity is also important for both the functioning of ecosystems and the generation of many other ecosystem services (e.g., Tilman and Downing, 1994; Tilman et al., 1996; Wood and Lenné, 1999; Loreau and Hector, 2001; Naeem et al., 1994). We focus on crop biodiversity and agricultural production. Crop biodiversity is shown to be critical in attempts to achieve food security. An emerging body of research focuses on the same research question, but using different methods reveals similar findings.

Evenson and Gollin (1997) provide evidence of the role of genetic diversity on agricultural yields. The role of biodiversity on productivity is also found to be positive and not negligible by Di Falco et al. (2007) and Smale et al. (1998). These findings are based on two different empirical approaches. The first is undertaken at an aggregate, i.e. multi-farm level, in which biodiversity is typically modelled as an input to the production process (e.g., Smale et al., 1998; Widawsky and Rozelle, 1998; Omer et al., 2007). However, the scale of these analyses does not allow one to control for individual farm characteristics, and such analyses implicitly assume that the underlying theoretical model can be scaled up to the macro level. The second approach focuses on the behaviour of individual farms using data collected from a cross section of farms. Although this overcomes the aggregation problem, it neglects changes through time (Di Falco and Chavas, 2009).

In using farm-level data from a cross-section of farms and collected at different points in time, a more recent paper by Di Falco et al. (2010) attempts to circumvent these shortcomings. The dataset, collected in 2002 and 2005, was derived from a survey of 1,500 farm households resident in the Central Highlands of Ethiopia, a drought-prone region with poor soils. Adoption of such a dataset enabled the researchers to overcome a number of statistical problems. The empirical strategy assesses the relationship between productivity, diversity, and rainfall, with results emphasizing the importance of *in-situ* biodiversity for food production.

Omer et al. (2007) adopted a different approach to empirically test the hypothesized positive relationship between biodiversity and levels of crop output. This analysis is based on a dataset of UK cereal farms for the period 1989–2000. Increases in biodiversity are shown to be critical in expanding the extent of what were thought to be the highest possible yields. A transition toward biodiversity conservation in some areas may be consistent with increasing crop yields in already biodiversity-poor modern agricultural landscapes. Smale et al. (1998) studied the relationships between crop biodiversity and wheat production in the Punjab of Pakistan. They find that genealogical distance and number of varieties are associated with higher average yields. The relationship between risk exposure and crop biodiversity has also attracted empirical attention. Exposure to risk is captured by variation either in crop yield or revenues. Widawsky and Rozelle (1998), using data from regions of China, find that the number of planted varieties reduces both the mean and the variance of rice yield. This finding is consistent with empirical research undertaken by Di Falco and Perrings (2005) and Di Falco et al. (2007).

Evidence of the narrowing of the genetic base of crops can thus indicate that farmers in the developing world are in fact becoming more and more vulnerable to environmental risk, such as the vagaries of weather.

10.4 LOOKING AHEAD: BIODIVERSITY AND DEVELOPMENT

Having established the scale and nature of biodiversity losses in section 10.2 and examined the critical role of biodiversity in the production of ecosystem services—by way of agricultural production—in section 10.3, this section reviews research that generates scenarios for future trends in biodiversity loss. These have implications for biodiversity policy and development (see section 10.5). For the most part, the proximate threats remain as documented by MA (2005) for the period 1950–2000: land-use and habitat change; over-exploitation; invasive alien species, pollution, and climate change. We begin by examining projections for forces that ultimately influence the proximate threats.

10.4.1 Underlying drivers

Underlying all the proximate threats already outlined are continued increases in the human population, rising incomes, and changing consumption patterns across the world. Between 1800 and 2011, the global, human population increased from one billion to seven billion, although growth rates have been falling in recent decades. This expansion is expected to continue for several more decades before reaching close to ten billion by 2050 (UN, 2011). A large literature emphasizes the threat to biodiversity and ecosystem functioning as a consequence of population growth (e.g. Cincotta and Engelman, 2000; McKinney, 2001; Harcourt and Parks, 2003; Balmford et al., 2001; Ceballos and Ehrlich, 2002).

McKee et al. (2003), for example, use data collected from a number of countries in order to examine the relationship between human population density and the number of threatened mammal and bird species. Their analysis shows that human population density and species richness play an important role in explaining numbers of threatened species. The model is then used to simulate predictions of numbers of threatened species in the future. On average, they are expected to increase by 7 per cent by 2020, and 14 per cent by 2050. While this might seem relatively low, the authors conclude that '[i]f other taxa follow the same pattern as mammals and birds . . . then we are facing a serious threat to global biodiversity' (p. 163). As with the EKC studies, however, some caution is in order when assessing empirical results based on small sample sizes and country-scale data. The nature of the analysis is highly prone to statistical problems that can bias the results.

With the expansion of the world's population, human societies have undergone a remarkable transition in which the majority of people now live in urban areas rather than rural ones. Much future population growth is thus expected to occur in urban areas; almost two billion new urban residents are expected by 2030, mainly in relatively small cities in developing countries (UN, 2012). Urbanization is expected to have significant effects on ecosystem services (e.g. Martine et al., 2008; Forman, 2008). Effects are expected both directly through the expansion of urban areas and indirectly through changes in consumption and pollution as people migrate into cities (McKinney, 2002; Liu et al., 2003; McGranahan and Satterthwaite, 2002).[5]

Grimm et al. (2008) discuss the direct effects of urbanization on biodiversity. First, within cities, urbanization impacts negatively on biodiversity, although not always. For example, a highly variable patchwork of habitats and human introductions of exotic species can actually boost urban biodiversity. Second, urbanization alters the composition of species due to human-induced changes such as altered temperatures, light, noise, and air pollution. Thus, urbanization can be a strong evolutionary force. Indirect effects are much more difficult to isolate empirically, although tentative evidence has begun to emerge. For example, DeFries et al. (2010) analyse satellite-based estimates of forest loss from 2000 to 2005 across forty-one tropical countries. They show that forest loss is positively correlated with urban population growth and exports of agricultural products. Their results highlight the importance of urban-based and international demands for agricultural products as potential drivers of deforestation. Again, however, note the dependence on small numbers of observations and coarse country-scale data.

10.4.2 Proximate drivers

Much research focuses on the issue of habitat conversion (driven by population and economic growth) as the key driver of biodiversity loss. This has been

[5] We note, however, that urban living can be associated with more 'environmentally friendly' modes of living due to people living in closer proximity, leading, for example, to less intensive per-capita energy usage and lower greenhouse gas emissions (see Glaeser, 2011).

incorporated by the MA (2005) indicators of the proximate drivers of biodiversity loss. Such drivers have intensified in recent years (see Butchart et al., 2010). The most important driver is habitat change (e.g. land-use change). These drivers are further influenced by the process of climate change. Although research typically focuses on single drivers of change, these are often related to one another. For example, land-use change can result in greater nutrient loading if land is converted to high-intensity agriculture, increased emissions of greenhouse gases (if forest is cleared), and increased numbers of invasive species (due to the disturbed habitat) (MA, 2005). The process of economic integration among countries can also play an important role in future biodiversity change. The number and distribution of species can indeed be affected by globalization of economic systems.

Looking ahead, the MA developed four plausible scenarios for ecosystems and human well-being. These explored two global development paths, one in which the world becomes increasingly globalized, and the other in which it becomes increasingly regionalized. In addition, they explored two different approaches to ecosystem management. The first approach comprises actions that are reactive: most problems are addressed only after they become obvious. In the second, ecosystem management is proactive: policies deliberately seek to maintain ecosystem services over the long term (MA, 2005). Under all four MA scenarios, the projected changes in drivers to 2050 result in significant growth in consumption of ecosystem services, continued loss of biodiversity, and further degradation of some ecosystem services. More specifically:

- Demand for food crops is projected to grow by 70–85 per cent, and demand for water by between 30 per cent and 85 per cent. Water withdrawals in developing countries are projected to increase significantly, although these are projected to decline in industrial countries (*medium certainty*).
- Food security is not achieved and child malnutrition is not eradicated (and is projected to increase in some regions under certain MA scenarios), despite increasing food supply and more diversified diets (*medium certainty*).
- A deterioration of the services provided by freshwater resources (such as aquatic habitat, fish production, and water supply for households, industry, and agriculture), particularly in the scenarios that are reactive to environmental problems (*medium certainty*).
- Habitat loss and other ecosystem changes are projected to lead to a decline in local diversity of native species (*high certainty*). Globally, the number of plant species is projected to be reduced by around 10–15 per cent as the result of habitat loss alone (*low certainty*), and over-harvesting, invasive species, pollution, and climate change will further increase the rate of extinction.

Predicting the response of biodiversity to climate change has recently become an active field of research (e.g. Dillon et al., 2010; Gilman et al., 2010; Beaumont et al., 2011; Dawson et al., 2011; McMahon et al., 2011). Although there is relatively limited evidence of current extinctions caused by climate change, research suggests that climate change could surpass habitat destruction as the biggest threat to biodiversity over the next few decades (Leadley et al., 2010). However, the multiplicity of approaches and the resulting variability in projections make it difficult to obtain clarity with respect to the future of biodiversity under different climate change scenarios (Pereira et al., 2010). Bellard et al. (2012) review both the ranges of possible impacts of climate change that operate at different scales, and the different responses that could occur at different levels of biodiversity. They show that species can respond to climate change challenges by shifting their climatic 'niche', for example, over time and space. While current estimates are highly variable, the majority of models indicate serious consequences for biodiversity. The worst-case scenarios suggest extinction rates that would qualify as the sixth mass extinction in the Earth's history.

10.5 BIODIVERSITY PROTECTION, POLICY, AND WELFARE IMPACTS

Biodiversity plays a crucial role in the production of a wide range of ecosystem services. In turn, these services often generate public benefits, which are rarely—if ever—considered by actors who profit from the exploitation of ecosystems. Hence, policies may be implemented in order to protect biodiversity. Both underlying and proximate drivers of biodiversity losses need to be addressed. Such policies, if effective, could play an important role in minimizing the probability of high extinction rates. In this section, we first examine the limited evidence for a relationship between biodiversity protection and economic growth at the country scale. We then investigate possible interlinkages at a smaller scale, focusing on the very low incomes of rural people in biodiverse developing countries. In the second part of this section, we review two different yet important policy instruments utilized to protect biodiversity: protected areas and bioprospecting.

10.5.1 Biodiversity protection, economic growth, and poverty

Given the possibility that more effective environmental regulation could help drive the increase in environmental quality at higher levels of income, Dietz and Adger (2003) examined the relationship between economic growth and biodiversity conservation using their multi-country dataset. Results suggest that the extent of government policy on biodiversity protection increases with economic development. More specifically, Dietz and Adger suggest that:

> low levels of income in a country may be correlated with restrictions on government enforcement of Convention on International Trade in Endangered Species (CITES) and other environmental legislation. (p. 30)

Within countries, however, there is little evidence that rising incomes have led to more biodiversity protection. Instead, where investment opportunities are limited to agriculture, increased incomes have been shown to result in greater conversion of habitat and thus biodiversity loss. For example, Zwane (2007) examined the relationship between income and forest clearing for households living in Peruvian tropical forest. She found income in the past to be positively associated with current deforestation levels. Due to the inability of people to obtain work outside agriculture, forest clearing is found to be associated with household labour availability. In conclusion, small increases in the incomes of the poorest were unlikely to reduce deforestation in the Peruvian context.

There is little doubt regarding the important role of ecosystem services in supporting the incomes and livelihoods of the rural poor in developing countries (MA, 2005; WRI, 2005; TEEB, 2010; World Bank, 2011). Turner et al. (2012) demonstrate how biodiverse areas supply important—and valuable—ecosystem services to the poor, even where the range of services considered is restricted to food, fuel, clean water, and so on. Yet these services are often insufficient to lift people out of poverty. In a recent special issue of the *Proceedings of the National Academy of Sciences*, 'On Biodiversity Conservation and Poverty Traps', the introductory paper by Barrett et al. (2011) identifies four classes of mechanisms that define the links between biodiversity conservation and poverty. More specifically, they focus on any self-reinforcing mechanism that causes poverty, however measured, to persist—i.e. poverty traps:

- Dependence of the poor on inherently limited natural resources in order to meet consumption needs;
- Vulnerabilities shared between the poor and biodiverse ecosystems—i.e. due to poverty, population growth, and environmental degradation;
- Failure of socio-political and economic institutions, including markets—e.g. labour markets, as in the study by Zwane (2007) summarized previously;
- 'Unintended consequences' and management failures as a result of decisions made over natural resource exploitation—e.g. the development of mineral resources leading to changes in watersheds that impact negatively on downstream communities.

Since one or more of these four mechanisms might apply in any given setting, the policy implications that follow may vary widely (Barrett et al., 2011). Thus, any starting point for policy design requires that 'careful site-specific diagnostics' are undertaken first (p. 13,910; see also Pfaff and Robalino, 2012).

10.5.2 Biodiversity policy and welfare impacts

Recent decades have witnessed widespread experimentation with different policies to conserve biodiversity around the world, from regulatory and command-and-control instruments, such as protected areas, to newer generations of voluntary and market-based instruments (see Albers and Ferraro, 2006, and Chapter 12 of this book by Miteva et al.). Regarding the latter, direct incentives for conserving biodiversity could be provided through payments for ecosystem services (PES). Alternatively, incentives could be supplied more indirectly. Either capital and labour could be redirected away from activities that underlie habitat loss such as certain types of agriculture, or commercial activities that supply ecosystem services could be encouraged via 'joint production' (Ferraro and Kiss; 2002; Bulte and Engel, 2006). Examples of the former include agricultural intensification and the development of off-farm labour opportunities; the latter include sustainable forestry, some non-timber forest products, and ecotourism. In this sub-section, we discuss the evidence of welfare impacts of two types of policy only: one command-and-control instrument, protected areas; and one market-based instrument, bioprospecting.

Protected areas

Dietz and Adger (2003) highlight the important role of protected areas in conserving biodiversity. It is the most common policy implemented to protect biodiversity in both developed and developing countries (MA, 2005). For example, in the most recent Global Forest Resources Assessment (FAO, 2010), national parks, game reserves, wilderness areas, and other legally established protected areas cover more than 10 per cent of the total forest area in most countries and regions. Yet, Myers (2003) reports that just over a third of the world's twenty-five biodiversity hotspots are found in protected areas. Typically implemented by government agencies, these areas have varying degrees of restriction with regard to their use, from 'strict nature reserves' through to areas in which some extractive uses may be permitted (IUCN, 2003). In principle, legal restrictions prevent human disturbance, thus contributing to the maintenance or recovery of ecosystem services (Ferraro et al., 2012).

In many developing countries, conserving biodiversity through the establishment of protected areas may impose an uncomfortable trade-off for policy-makers: people often rely on natural resources within these areas and human uses may not be compatible with biodiversity protection (Albers and Ferraro, 2006). However, *effective* protected area management requires that governments have the ability to enforce protection, which is often not the case.[6] Local people are often dependent on natural resources in protected areas for incomes and livelihoods, which may lead to conflict with governments (Engel et al., 2013). One consequence of such conflicts is that there may be incentives for local people to overharvest resources without regard to the costs imposed on others.

Where protected areas are effectively enforced, people's access to resources might be restricted. Patrols and fines for illegal extraction can impose high costs on local people (Bulte and Engel, 2006). Much research on the socio-economic impacts of protected areas in developing countries tends, however, to show little more than 'that protected areas are established near poor people and provide both opportunities and constraints to economic development' (Ferraro et al., 2012, p. 35; see also Coad et al., 2008; Wilkie et al., 2006).[7] Barrett et al. (2011) highlight a number of challenges for studies attempting to better understand the relationship between the protection of ecosystems and people's well-being—for example, the absence of historical data (see also Chapter 12 of this book by Miteva et al.).

A recent, growing body of empirical research appears to support the economic intuition underlying the trade-off between biodiversity protection and welfare (Andam et al., 2010; Sims, 2010; Barrett et al., 2011). Specifically, this research applies certain methods known as 'programme evaluation techniques'. These include randomized field experiments and statistical analyses, which assess the environmental effectiveness and welfare impacts of policies to conserve biodiversity. Such techniques are applied in order to deal with the fact that biodiversity policy interventions are commonly implemented in a non-random manner, over both time and space. Protected areas, for both political and economic reasons, are often situated in areas with few profitable, alternative uses (Albers and Ferraro, 2006).

A number of examples of recent research on the welfare impacts of protected areas can be found in the special issue 'On Biodiversity Conservation and Poverty Traps', already referred to. First, Ferraro et al. (2011) estimate the impacts of protected areas on poverty and deforestation across diverse sites in Costa Rica and Thailand. Little evidence is found that protected areas trap historically poorer areas in poverty. Yet, the characteristics associated with the

[6] Where protected areas are effective in conserving biodiversity, their impacts may, however, be diminished by the displacement of extractive activities to unprotected areas nearby (e.g. see Andam et al., 2008).

[7] Although outside the scope of this review, Ferraro et al. (2012) also note that there are no studies that have examined the cost-effectiveness of protected areas.

most poverty alleviation are not always the ones associated with the slowing-down of deforestation. Second, in Uganda, Naughton-Treves et al. (2011) found that Kibale National Park protected both forest and primates. There is no evidence that the Park was a poverty trap. Third, McNally et al. (2011) analyse the impacts of Saadani National Park on local households in Tanzania. By restricting access to mangrove timber, the Park increased people's incomes from fishing and shrimping, as well as their indirect benefits as a consequence of mangrove protection.

Bioprospecting

There can be private benefits from actions that lead to biodiversity protection (Albers and Ferraro, 2006). For example, non-governmental organizations such as The Nature Conservancy (TNC) depend on voluntary contributions in order to provide biodiversity through the acquisition of habitat, both in developed and developing countries. One highly debated source of private incentives for biodiversity protection is bioprospecting.

As noted earlier, bioprospecting is the search for valuable compounds from wild organisms. This involves searching for, collecting, and deriving genetic material from samples of biodiversity that can be used in commercialized end-products. It has been touted as a mechanism for both discovering new pharmaceutical products and saving endangered ecosystems via the financing of conservation. For example, Rausser and Small (2000) claimed that the value of protecting certain ecosystems for bioprospecting can be quite high. Given that the annual market size for products based on genetic resources has been estimated to lie within the range of US$220–300 billion (Deke, 2008), there would appear to be strong enough private incentives for protecting biodiversity through bioprospecting.[8]

This view, however, has been challenged in a seminal paper by Simpson et al. (1996). In this paper, both the marginal benefits and costs of conservation are addressed from a biosprospecting angle. In other words, the authors consider the costs and benefits of protecting one additional species. The marginal value is, however, decreasing due to redundancy: there are many species that may perform the same function. The total economic cost of losing some species is also included, which is found to be negligible for both high and low levels of species. This, of course, creates a problem for incentives to conserve habitats for the protection of biodiversity.

Contracts have been negotiated between pharmaceutical firms and the government, or individuals who control biodiverse ecosystems. Yet the

[8] This figure is based on sales in world markets for products sold in the healthcare (e.g. pharmaceuticals, cosmetics), agriculture (e.g. seeds, crop protection), and 'other biotechnology' (e.g. bioenergy) sectors (see Deke, 2008).

number of private partnerships remains small. For example, the National Biodiversity Institute (INBio) of Costa Rica negotiated a contract with Merck in 1989 (Sedjo, 1992). In this, US$1 million was paid to INBio for rights to screen plants for useful chemicals over two years. INBio was to receive royalties in the event of any successful commercial applications. In general, however, there is relatively little empirical information regarding these kinds of transactions (Deke, 2008).

While doubts have been raised about bioprospecting's potential to both effectively conserve biodiversity and alleviate poverty, very few studies have assessed both the socioeconomic and environmental impacts in an empirically robust manner (Barrett et al., 2011). Yet, the socioeconomic impacts in developing countries are likely to be afflicted by the same kinds of problems as discussed for protected areas. In particular, property rights to genetic resources may be difficult to specify and even more difficult to enforce. Furthermore, there may be missing legal frameworks for regulating benefits sharing from genetic resources (Dhillion et al., 2002), and excessive bureaucracy may divert finance for R&D away from the conservation of *in-situ* genetic material (ten Kate and Laird, 2000).

10.6 CONCLUSIONS

According to Barnosky et al. (2011), the recent loss of species, while dramatic, does not yet qualify as a 'mass extinction', at least from the perspective of the Earth's fossil record. Only a few per cent of assessed (i.e. known) species have been lost in recent decades, although species may have been lost that had never been recorded in the first place. Yet, there are clear indications that losing species currently in the 'critically endangered' category could lead to a mass extinction on a scale that has occurred only five times in the previous 540 million years. If so, another mass extinction could occur within 'a few centuries' (Barnosky et al., 2011, p. 56). Thus, it would seem that biodiversity faces a bleak future, at least in the long run, given the projections of proximate and underlying drivers of change presented in section 10.4. Although many recent reviews and projections point in this direction, we note that projections of biodiversity loss do vary widely according to the assumptions and data used in the models. Further research is required into biodiversity's role in ecosystem resilience and in providing a kind of buffer against reaching a 'tipping point' in system dynamics (see Chapter 2 of this book by Helm and Hepburn). Ecosystem collapse could potentially speed up rates of biodiversity loss.

Recent years have witnessed intensifying attention from policy-makers in response to the documented decline of biodiversity. There has been a realization that the need for greater efforts to conserve biodiversity is due to the fact that the benefits of biodiversity cross national borders. This implies a need both for international institutions that can govern biodiversity effectively, and a continued focus on improving policy design for effective biodiversity protection. Indeed, Butchart et al. (2010) note recent improvements in indicators of attempts to counter biodiversity losses. These include the extent and biodiversity coverage of protected areas, sustainable forest management, and policies to deal with invasive species. However, even where such policy responses are shown to be effective—still a relatively rare process, as discussed for protected areas and bioprospecting in section 10.5—they still often fail to benefit the rural poor in developing countries. And where policies restrict people's access to and use of natural resources, they could even be made worse off as a consequence of policy implementation.

Policy design and implementation are further complicated by the high degree of uncertainty regarding our knowledge and understanding of biodiversity and its interlinkages with ecosystem services. As illustrated with the case of agricultural biodiversity in section 10.3, progress on research is being made and sensible policy responses have been forthcoming. Maintaining diverse plant varieties in farmers' fields, *in-situ* conservation, vis-à-vis storing germplasms in gene banks, for example, is increasingly regarded as an effective way of conserving plant biodiversity (Benin et al., 2004; Bezabih, 2008). At the heart of whether *in-situ* conservation could be pursued as a fruitful strategy is whether it generates benefits for farmers. Here, policies such as PES could provide the necessary incentives for local conservation activities that yield wider social benefits.

An emerging body of empirical evidence appears to suggest, however, that individual policies or strategies (e.g. protected areas or PES) may be unable to reconcile both biodiversity conservation and poverty alleviation objectives in some settings (Barrett et al., 2011). For instance, protecting biodiversity from invasive species may require both monitoring and eradication activities in existing protected areas (Albers and Ferraro, 2006). Siting of areas, and the management of buffer zones among areas and their surroundings, could reduce the opportunities for invasive species to take hold. Similarly, climate change impacts may be mitigated to some degree if the siting of protected areas and the management of nearby land and wildlife corridors allow species to move and adapt gradually to changes in climate. While some of the activities connected with these policy strategies such as eradication or management efforts could potentially generate local employment and other, associated benefits, they are unlikely to have a broader impact on poverty. To achieve this requires policies, separate from biodiversity conservation, that actually identify and tackle the root causes of poverty. If, on the other hand, these are in some way related to natural resource use and dependence then there may be little option but to carefully design policies that attempt to address both the loss of biodiversity and poverty.

Assuming that policies can be designed to effectively conserve biodiversity and, at the minimum, 'do no harm' to the poor, there remains the question of how such policies might be financed. Given ever-tightening constraints on public expenditures in many of the world's developed economies, attention is shifting to alternative sources of finance for biodiversity protection. Recently, there has been much speculation about the role of Reducing Emissions from Deforestation and Degradation (REDD +) in financing the preservation of tropical forests rich in both carbon and biodiversity (e.g. Gardner et al., 2012). Although REDD + was originally posited as a strategy primarily to mitigate the effects of climate change, a series of 'safeguards' relating to biodiversity protection and the livelihoods of the poor were adopted by parties to the United Nations Framework Convention on Climate Change at Cancun in 2010. In principle, REDD + could enable a number of changes to the governance and use of forests, leading to increased levels of biodiversity protection; for example, *in-situ* conservation through the establishment of new protected areas and associated corridors for connecting landscapes. Yet, mechanisms for the long-term financing of REDD + remain uncertain. Also, policies for operationalizing REDD + will need to overcome the same challenges that have bedevilled the management of natural ecosystems in recent decades. As the world's population grows and becomes increasingly urbanized, our dependence upon biodiversity for our incomes and livelihoods is unlikely to diminish. Stemming biodiversity loss is thus critical for ensuring our future prosperity. However, this requires not only a willingness to pay for biodiversity protection, but also that we actually pay for biodiversity, and in the process implement effective policies on the basis of learning from past policy experiences—for good or ill.

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11

Regulating Global Biodiversity: What is the Problem?

Timothy Swanson and Ben Groom

11.1 INTRODUCTION

Regulating *global* biodiversity fundamentally concerns the question of how much total habitat conversion we wish to undertake across the globe as a whole (Swanson, 1994). This is a distinct question from how each individual state might view the importance of conservation for its own purpose, or even for the benefit of others. Our question is focused on the impact of the aggregate level of land-based development on Earth. This is a global question—from the sustainability perspective—concerning whether the Earth's system can continue to extend highly productive land uses (such as agriculture and hence the associated human population). It is very similar to the questions raised in general at the 1992 Rio de Janeiro United Nations Conference on Environment and Development (UNCED)—how is global development to be made consonant with the global environment? In this chapter we examine the basic structure of the problem that is being addressed in this context, and the range of *international* policies that have been attempted. Finally we examine why these policies have been less than wholly successful.[1]

The chapter is arranged as follows. Section 11.2 sets out the stylized facts regarding development and biodiversity—what is the global problem that needs to be regulated? Section 11.3 presents the biodiversity regulation problem as a simple land-use model of North–South interdependence in the biotechnology

[1] We are analysing the problem as one of halting global conversion, and then determining the incidence of such a policy, given that different countries have experienced more conversion than others (for a similar analysis in the context of the Montreal Protocol, see Mason and Swanson, 2003). We argue that the fundamental problem of global biodiversity is the incidence of a policy halting conversion at this juncture, and the perception of the relevant states concerning the unfairness of that incidence.

sector with biodiversity as a global public good. The problem of global surplus division is discussed in relation to some simple solutions from cooperative bargaining theory. This illustrates the essential factors that determine the co-operative solution within this framework. Section 11.4 discusses the international policies for addressing this problem, the Convention on Biological Diversity (CBD), and associated institutions, in light of the essential factors highlighted by bargaining theory. Section 11.5 discusses why there are no lasting solutions in place for the global biodiversity problem. A conclusion follows.

11.2 WHEN IS BIODIVERSITY REGULATION A GLOBAL PROBLEM?

The regulation of global biodiversity concerns management of the ongoing practice of land conversion across the globe. This is a practice that commenced about 10,000 years ago, and has been targeted first at some continents and then at others. The first places to experience massive land-use change have been the more temperate areas (Europe, North Asia, North America), and in some cases the land conversion that occurred long ago is near complete.[2]

This has resulted in some striking asymmetries on Earth. For one thing, the parts of the Earth where the vast majority of biodiversity resides are few. The majority of species on Earth are now believed to reside in the final three great tropical rainforest systems (Amazon, Congo, Indonesian). And most indicators of species' continued existence indicate the same general locations and nations as the hosts of the remaining diversity (see Table 11.1).

On the other hand, a quick look at the same states concerned indicates that there is an interesting but inverse correlation between species richness and other forms of wealth. Many of the states that are amongst the wealthiest in terms of biodiversity are also amongst the poorest in terms of standard measures of income (see Table 11.2).

This asymmetry between the holders of biodiversity assets and those holding other forms of assets demonstrates one of the basic problems of managing global resources—the asymmetry in endowments. It makes outcomes difficult to negotiate, when starting points are so far apart.

How did this asymmetry result? The fundamental cause is the order in which states have converted their lands. Some did so thousands of years ago (Europe), others hundreds of years ago (North America), and others have done so over the past few decades (Latin America, Southeast Asia). The globe has lost 4 per cent of its forested lands—to agriculture—over the past couple of

[2] For example, the amount of wilderness habitat (defined as unaltered land mass of at least 2,500 sq. km) in Europe is now certifiably zero (World Resources Institute, 2003).

Table 11.1. Countries with greatest 'species richness'

Mammals	Birds	Reptiles
Indonesia (515)	Colombia (1,721)	Mexico (717)
Mexico (449)	Peru (1,701)	Australia (686)
Brazil (428)	Brazil (1,622)	Indonesia (600)
Zaire (409)	Indonesia (1,519)	India (383)
China (394)	Ecuador (1,447)	Colombia (383)
Peru (361)	Venezuela (1,275)	Ecuador (345)
Colombia (359)	Bolivia (1,250)	Peru (297)
India (350)	India (1,200)	Malaysia (294)
Uganda (311)	Malaysia (1,200)	Thailand (282)
Tanzania (310)	China (1,195)	Papua New Guinea (282)

Source: McNeely et al. (1990).

Table 11.2. GDP per capita in the diversity-rich states (PPP)

Country	2011 GDP per capita	Country	2011 GDP per capita
Tanzania	$1,336	Papua New Guinea	$2,363
Uganda	$1,188	Indonesia	$4,094
India	$3,203	Bolivia	$4,503
Ecuador	$7,655	Colombia	$10,279
China	$7,418	Brazil	$11,640
World average	$10,071		
OECD average	$30,371		

Source: World Bank Development Indicators 2012.

hundred years. At the same time many of those countries that have deforested have also advanced their agriculture and other industries dependent on larger populations and urban densities. In the long view, development has often been initiated with land conversion and agriculture. For this reason, it has long been the case that national incomes have gone up while forested areas have gone down.

From this perspective it is possible to view the problem of regulating global biodiversity as one of the regulation of global land-use conversion, where the external costs of conversion are increasing as the conversions continue apace. In this framework the concern over sustainability is that there may be a global—or aggregate—limit to the amount of conversion that might be able to be incurred.[3] Figure 11.1 gives a depiction of a regulatory scenario for this

[3] We are defining the global problem of biodiversity as that problem that requires international regulation for its resolution. Many other facets of the biodiversity decline may be addressed through appropriate domestic regulation (e.g. watershed management) or bilateral transfers (e.g. payments for parks and protected areas).

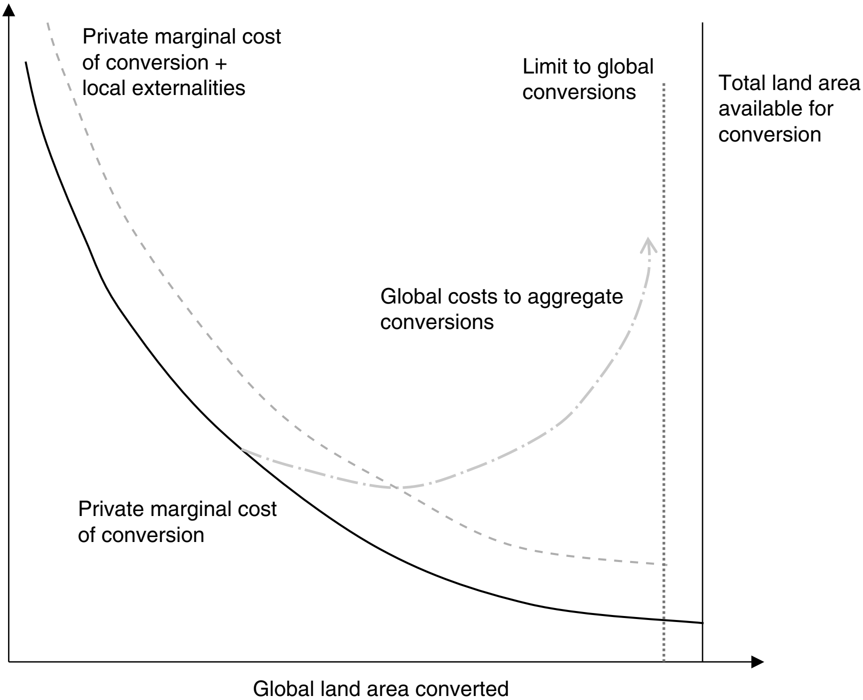

Fig. 11.1. Global regulation of biodiversity: optimal land-use conversion

problem of global land-use conversion. It is a problem of the un-internalized costliness of land conversions.

In Figure 11.1, the various types of biodiversity problems are segregated: local, regional, and global.

The *lowermost curve* in the diagram represents the 'perceived marginal cost (MC) of land conversion'—as viewed from the perspective of the converting landowner (most often the state or private landowner with jurisdiction over the land).[4] The *dashed line* just above this curve concerns the more generalized or social marginal cost of such land conversions, when the role of that parcel of land is considered as part of a larger ecosystem. This social cost represents the local and regional externalities flowing from that particular piece of terrain being converted.

These local and regional costs may not be fully internalized to the decision-maker considering land conversion, because they might flow to the broader watershed community (who receive clean water from the unconverted

[4] We show MC as declining over the entire range of land conversion, but the same points (about various forms of externalities) stand if the MC begins to incline at some point in the process.

watershed), or the broader forest community (who may desire a wider range of uses of the land concerned), or even the broader global community (who may hope for large unconverted land areas to support charismatic species, such as the panda or the elephant). These are the problems considered by groups who hope to internalize local and regional values of ecosystems to existing decision-makers (TEEB, 2010). The conversion of any piece of land will involve some amount of externalities (given its role in other systems), but it will not matter much in which order the land is converted; the externalities are borne whenever that piece is lost. For this reason the external cost is represented by a simple vertical shift of the marginal cost curve upwards (a constant cost to any piece of land converted, whenever it is converted).

The problem of regulating global biodiversity is different from this 'local externality' problem, and concerns the potential limits to a particular development strategy—here the practice of land conversion (Swanson, 1995*b*). In Figure 11.1, this is represented by the uppermost marginal cost curve—where the social marginal cost of continued conversion (beyond some limit) goes to infinity.[5] This rapidly escalating cost of conversion would be the case if there is indeed a limit on the total amount of global land-use conversion that is feasible—while retaining a relatively stable and resilient biological system that is capable of maintaining a life system within which humans can survive.[6]

Is sustainability really a problem in this context? There are several reasons why biodiversity may be critical to maintaining the entire production system. First, unconverted lands act as 'firebreaks' that reduce the rate of arrival of new biological problems (Goeschl and Swanson, 2003*a*). When such problems do arise, it is recognized that genetic resources play a crucial role in supplying the options or solution concepts within the life sciences industries (Kassar and Lasserre, 2004). Biodiversity does this by supplying genetic resources to R&D sectors supplying the life sciences industries (Sarr, Goeschl, and Swanson, 2008). It may be possible for technological advance to substitute for biodiversity resources in the long run, but at least at present (and certainly in the past) most problems in the life sciences were dealt with using existing genetic resources (Swanson, 1995*a*).

Figure 11.1 also demonstrates the difficulty involved in halting the global land-use conversion process. States and private landowners perceive a marginal cost that enables conversion to take place—and the converted receive an uncompensated flow of benefits from the unconverted, providing further

[5] Bringing to mind the comment attributed to Michael Toman (Toman, 1998) on Costanza et al. (1997) that their estimate of natural capital's aggregate value at $50 trillion represented 'a very serious underestimate of infinity'.

[6] The global problem has more in common with the problem of climate change than the problem of ecosystem valuation. It is a question of determining whether there is a limit to conversion-based development.

incentive to join the ranks of the converted.[7] If this process continues, the converted system continues to become more unstable and less resilient (as biological problems are a function of scale and contiguity), while the area of unconverted lands (from which solutions must originate) becomes smaller (Goeschl and Swanson, 2003*a*). For this reason it is to be anticipated that there is a limit to this process of conversion, after which the cost of ongoing conversion goes to infinity, representing the instability of ongoing aggregate conversions.

11.3 STRUCTURE OF THE BIODIVERSITY BARGAINING PROBLEM: THEORY AND CASE STUDY

As we have argued earlier, there is considerable value in global biodiversity. There are many reasons to think that the conversion of biodiverse lands is greater than would optimize global welfare. Part of the problem is that there exists a fundamental asymmetry in complementary endowments: broadly speaking, between a biodiverse 'gene-rich' South, and a 'value' and 'technology-rich' North. In order to realize the global value it is necessary for both parts of the world—converted 'gene-poor' North and unconverted 'gene-rich' South—to cooperate in both the allocation of these inputs and the subsequent sharing of rents.

The rents may accrue generally as use or non-use values by and large in the North. Yet deeper mutual interdependence can be found in the biotech sector in agriculture and pharmaceuticals. For example, Gatti et al. (2011) focus on agriculture, in which the production of agricultural innovations is dependent on joint production using the two complementary yet asymmetric endowments: (i) human capital in the North; and, (ii) genetic material from biodiverse lands in the South. The North produces agricultural innovations that are beneficial to both the North and the South, using endowments from each. The general problem is that the North generates a final product, which can benefit both parties, but does so in reliance in part on the maintenance of a natural habitat sector in the South.

Ultimately, some sort of bargain must be struck in order to share the value of the product, in recognition of the joint production that is occurring. This is the fundamental problem we now examine: how should the global surplus be distributed in order to allow joint production (cooperation) to proceed? This is the *biodiversity bargaining problem*.

[7] This is the case because the unconverted lands reduce the arrival of problems and provide solutions—but the benefits from these activities are realized by reason of increased production on converted lands.

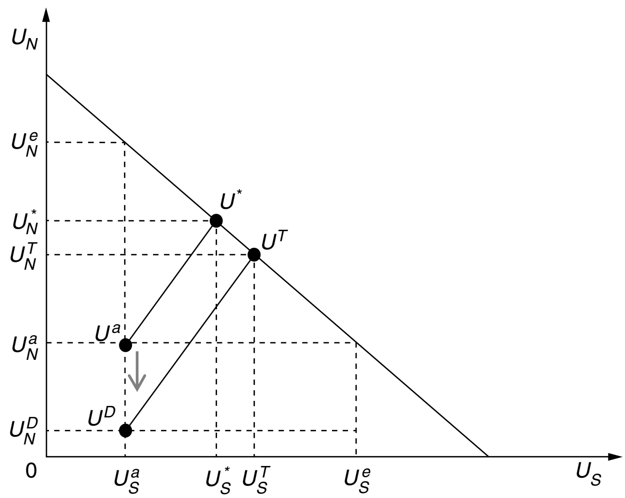

Fig. 11.2. A bargaining game—defined by conflict point and cooperative frontier
Source: Gatti et al. (2011).

We now place the biodiversity bargaining problem into the general bargaining structure following Nash (1953) and Rubenstein (1981). Figure 11.2 shows the essential problem. The axes measure the outcomes for two agents, here the North and South, in terms of 'utility': U_N and U_S. The conflict point, given by point U^a, represents the outcomes in the absence of cooperation. Payoffs along the bargaining frontier can be attained only if resource allocations are efficient. This limit of cooperative possibilities is shown by the line running between the two rational extremes (U_S^a, U_S^e) and (U_N^a, U_N^e). Rational agents will not accept any bargaining outcome with a lower payoff than at the conflict point (U_N^a, U_S^a), since non-cooperation is always available to them. The distance between the conflict point and the bargaining frontier measures the gains from cooperation. For example, in Figure 11.2, given a point on the bargaining frontier, U^*, the cooperative gains are $U^C = U^* - (U_N^a + U_S^a)$.

In the biodiversity bargaining game the conflict point describes an autarkic state of non-cooperation in which there is no exchange of biotechnological outputs (e.g. new plant varieties) and no transfer of surplus to coerce or contract land allocations. In the agricultural example from Gatti et al. (2011), the South benefits neither from biodiverse lands, which is locally unproductive residual land, nor from improved agricultural technology. The North gains only from spillovers from residual unconverted land, which is typically a public good. The South is not remunerated for its endowments, and production in the North is at the mercy of residual land allocations in the South.

The bargaining frontier is the set of outcomes when the problem of cooperation is solved as if by a single vertically integrated industry, in which the issues of sovereignty, jurisdiction, and asymmetry are ignored. Given the complementary nature of the endowments, the globally efficient solution results in more southern land being allocated to reserves land than under autarky (e.g. Gatti et al., 2011). When acting cooperatively the South becomes more specialized in reserves and the North specializes in R&D and final production.

The fundamental problem is that the bargaining solution is indeterminate. A solution might distribute surplus in accordance with U^{*} in Figure 11.2, but it could just as well be anywhere else on the bargaining frontier, or indeed the interior. One family of solutions to a cooperative bargaining game is the Nash bargaining solution. In the Nash Cooperative Bargaining Game (NCBG), there are primarily two determinative characteristics of the outcome: (a) the parties' respective conflict points; and (b) the parties' respective bargaining powers. The key insights from the Nash bargaining solution are firstly that rational agents will agree on some point on the bargaining frontier such as U^{*} in Figure 11.2; and secondly, the specific solution depends on the bargaining power of the respective parties. If bargaining power is not determinate then the solution to the NCBG is an indeterminate point on the bargaining frontier.[8]

Bargaining strength may derive from many factors that are characteristic of the agents. Typically, this is reflected in a 'sharing rule' or norm that determines the share of the overall cooperative surplus that will accrue to each party. Various norms have been adopted in international negotiations over rivers, fisheries, and other resources (Barrett, 2002).[9] The main point is that the 'sharing rule' must be accepted by both parties (or able to be imposed by one) in order to be a lasting resolution. We will term this a 'fair' resolution to the bargaining game (and return to this discussion in section 11.5). Importantly, the more bargaining power a party has, the more of the cooperative surplus that party will command.

The Nash Bargaining Solution is an axiomatic solution to the indeterminacy of the bargaining problem. Nash (1953) showed that the unique solution to the bargaining problem is the Nash solution when the following axioms hold: (i) invariance: the preference ordering of the parties are invariant to linear transformations; (ii) Pareto optimality: all gains to bargaining are exhausted;

[8] Formally, the general solution to an asymmetric NCBG is given by the maximization of: $(U_N - U_N^a)^{\alpha}(U_S - U_S^a)^{(1-\alpha)}$s.t. $U_N + U_S = U^{*}$, where the parameter $\alpha \in [0, 1]$ is an index of relative bargaining power. The outcomes of this bargaining problem for the North and the South are then: $U_N^{*} = (1-\alpha)U_N^a + \alpha(U^{*} - U_S^a)$ and $U_S^{*} = \alpha U_S^a + (1-\alpha)(U^{*} - U_N^a)$. See, for example, Nash (1953).

[9] See, for example, Bowles and Gintis (2000), who discuss norms in the context of the ultimatum game. Here, large deviations from a 50:50 split are frequently rejected. There is also evidence to show that sharecropping agreements frequently take the form of 50:50 shares (e.g. Bardhan and Singh, 1987).

(iii) independence of irrelevant alternatives (IIA): the ordering of preferences over two possible solutions is not influenced by the introduction of a third alternative; and (iv) symmetry: if the position of the players is symmetric, then each player should be treated symmetrically. The axioms essentially embody fairness (via symmetry) and some procedural elements (via IIA) and rationality (via Pareto optimality). Therein lies the appeal of this bargaining solution.

To uphold any solution requires the conclusion of some sort of a contract between the North and the South, and the agreement of its terms. With regard to a global public good like biodiversity, such contracts could be enshrined in an international environmental agreement. These would specify the efficient land allocations and contain transfers of global surplus, which reflect the agreed sharing rule and place the parties at a point on the bargaining frontier. One extreme possibility is that the agreed share is zero for one of the parties. In this case, 'extreme point' contracts arise. These special cases would place the South at point (U_N^e, U_S^a), or the North at point (U_N^a, U_S^e), and result when one party is devoid of all bargaining power. Such contracts leave one party no better off than in conflict, while the other accrues the entire global surplus of joint production, despite both parties contributing essential inputs.

It is important to realize that all such contracts are 'efficient' in the sense that they allow the agents to attain the bargaining frontier. It is obvious, however, that not all international negotiations are approached and resolved by reference to a rational bargaining process, or lead to an obvious solution to the bargaining game. So, although any point on the bargaining frontier is indeed an efficient solution to the game, it might not be a 'fair' one in the sense of achieving a lasting resolution to the bargaining problem.

Clearly, international agreements ought to be striving to coordinate or coerce parties to cooperate and move towards the bargaining frontier. The bargaining framework shows that this requires an acceptable and fair sharing rule for the global surplus. With this framework in mind, we now analyse some solutions offered by the main international agreements on biodiversity.

11.4 ADDRESSING THE BIODIVERSITY BARGAINING PROBLEM: INTERNATIONAL POLICIES

11.4.1 The Convention on Biological Diversity and National Sovereignty

The main international agreement concerning biodiversity conservation is the CBD. The issue of providing for shares is a fundamental principle of the framework convention. Indeed, 'benefit sharing' is the third objective of the CBD:

> the fair and equitable sharing of the benefits arising out of the utilisation of genetic resources, including by appropriate access to genetic resources and by appropriate transfer of relevant technologies, taking into account all rights over those resources and how benefits are shared based on a set of agreed norms and principles derived from ethics and equity. (UNEP, 2008)

In addition, the preamble of the CBD makes clear that genetic resources represent the sovereign resources of individual states, and removes any question regarding the possibility of unlicensed expropriation or global free-riding. The preamble provides that domestic regimes have absolute sovereignty over their genetic resources. Article 3 provides that 'states have . . . the sovereign right to exploit their own resources pursuant to their own environmental policies'. Article 9 provides that any use of or access to a state's domestic resources must be in accordance with the principles of informed consent and equitable benefit sharing. Hence, the first point to make is that the CBD does emphasize at its core the importance of addressing and resolving the biodiversity bargaining problem.

11.4.2 An international fund mechanism for biodiversity?

The CBD also addresses the question of the mechanism by which this sharing is to be accomplished. This is provided for in Sections 2 and 4 of Article 20 of the CBD:

> 2. The developed country Parties shall provide new and additional financial resources to enable developing country Parties to meet the agreed full incremental costs to them of implementing measures which fulfil the obligations of this Convention . . .
>
> 4. The extent to which developing country Parties will effectively implement their commitments under this Convention will depend on the effective implementation by developed country Parties . . .

The mechanism by which such transfers are to occur is also indicated under the terms of the Convention. It further provides in Section 1 of Article 21 that:

> [t]here shall be a mechanism for the provision of financial resources to developing country Parties for purposes of this Convention on a grant or concessional basis the essential elements of which are described in this Article.

These provisions of the CBD create the potential for a financial mechanism by which North–South transfers might occur. To some extent, this mechanism has come into existence through grants under the Global Environment Facility (GEF), but no independent 'green development mechanism' has yet to come into existence (King, 1994). To this point transfers under the aegis of the CBD continue on a more ad hoc basis.

More crucially for our purposes, the motivational principle under this part of the Convention is about compensation of costs. This approach misconceives the basic nature of global public good provision. The CBD describes the reason for payments to those states providing biodiversity services as one of compensation for burdens undertaken, not of the sharing of surplus generated. In the next section we provide the reasons why this is not the correct approach to the problem of biodiversity regulation.

11.4.3 Incremental costs contracting: an 'extreme point' contract

The contractual solution applied to the biodiversity bargaining problem can be found under the terms of the CBD and its financial instrument, the GEF, in the form of the concept of *incremental costs (IC):*

> [the North] shall provide new and additional financial resources to enable *[the South]* to meet the agreed full incremental costs to them of implementing measures which fulfil the obligations of this Convention. (Art. 20, CBD)

The meaning of the term 'incremental costs' is further defined within the founding instrument of the GEF as:

> [the costs of] additional national action beyond what is required for national development [the baseline] that imposes additional [or incremental] costs on countries beyond the costs that are strictly necessary for achieving their own development goals, but nevertheless generates additional benefits that the world as a whole can share . . .[10]

So, where does the IC contract place the negotiating parties in the bargaining set? In terms of the preceding analysis, the IC contract requires the North to compensate the South for the additional costs it incurs by electing the cooperative development path rather than its baseline development strategy.[11] There is no allusion to or provision for enhanced sharing by the South in the cooperative surplus by reason of this election, but only provision for the compensation of its costs incurred to generate additional benefits that the world as a whole can share. Importantly, neither does the contract condition payment on the level of the South's reserves.

In short, the IC contract does not bear any of the hallmarks of the efficient contract that would be anticipated to arise out of a resolution of the NCBG. Instead, it is a straightforward offer of the extreme point contract, in which the North offers the South compensation for its costs incurred in participating in the cooperative outcome. In terms of Figure 11.2, the IC

[10] GEF/C.7/Inf.5: para.2 & GEF/C.2/6 para.2; see King (1994).
[11] In terms of the model, choosing the efficient land allocation, t^*, rather than t^a.

contract places the parties at point (U_N^*, U_S^e), in which the North receives the entire global surplus.

Of course, the IC contract is, on the face of it, cost-effective. That is, it appears to obtain the biggest 'bang for the buck' since the North pays the lowest possible level of compensation to the South. The question for analysis is whether such a bargain—albeit efficient—can indeed be a final resolution to the biodiversity bargaining game. We return to this question in section 11.5.

11.4.4 Access rights and access and benefit sharing (ABS): can property rights solve this?

As mentioned earlier, the third objective of the CBD is to ensure benefit sharing in accordance with some international norms, and Article 9 provides that any use of or access to a state's domestic resources must be in accordance with the principles of informed consent and equitable benefit sharing. Article 15 provides for the idea that traditional knowledge and information is to be compensated. The Bonn Agreement of 2004 outlines mechanisms and instruments (such as up-front payments, revenue-sharing rules, and royalties) that can be used to facilitate benefit sharing. It is widely agreed that these mechanisms are very much in their infancy in terms of efficacy (UNEP, 2008).[12]

This is a private or market-based approach to creating a negotiated solution to the bargaining problem.[13] North and South can solve this problem at many different levels, one of which might be through negotiations between private firms in the two spheres. Such an approach hinges upon the agreement of a transaction regarding joint production, based on property rights transfers between each (Sarr and Swanson, 2011).

The basic difficulty with a property rights-based resolution is that there are no agreed property rights at the international level with regard to natural biological materials—and this means that firms in the South have no foundation from which to negotiate. To obtain internationally recognized property rights to the information contained within biological materials it is necessary to either improve them, or at a minimum to demonstrate the scientific process or method by which they may be used to generate an innovation.[14]

[12] Too often a genetic access regime is more like a legal checklist than a licensing agreement.

[13] In general there is nothing inefficient about having private bargaining determine the distribution of benefits resulting from the achievement of the socially optimal outcome *à la* Coase (Coase, 1960).

[14] The CBD creates an internationally recognized right to the physical genetic resources themselves, but is silent on the question of the informational values that originate from such resources. Since information flows freely, it is a relatively straightforward matter to become acquainted with that information without the transfer of physical materials themselves. (It is akin to becoming acquainted with the recipe, without having to take possession of the cake itself.) A large amount of effort has been expended on the creation of analogous rights in information from purely genetic resources from agricultural plant breeding—so-called Plant Breeders'

Without a recognized property right, bargaining cannot commence (Sarr and Swanson, 2011).

So, for these reasons, it remains difficult to initiate any sort of private bargaining over joint production with genetic resources. This will be the case so long as rights in the informational values of natural capital are non-existing (Swanson, 1995*a*). Even if these rights are established, the private approach to bargaining must necessarily remain only a partial solution. While these private values are thought to be significant, they do not capture the full social value of the stock of genetic resources arising from its ability to overcome well-known phenomena associated with pathogen adaptation and resistance.[15]

11.4.5 Whatever next? The Nagoya Protocol on Benefit Sharing

Most recently, at the 2010 CBD Conference of the Parties held in Nagoya, the parties proposed the text for a new Protocol on Access to Genetic Resources and Benefit Sharing (the Nagoya Protocol). This Protocol makes more explicit many of the terms previously contained within the Articles of the Convention.

For example, the Nagoya Protocol Article 5 on Fair and Equitable Benefit Sharing provides in part as follows:

> In accordance with Article 15, paragraphs 3 and 7 of the Convention, benefits arising from the utilization of genetic resources as well as subsequent applications and commercialization shall be shared in a fair and equitable way with the Party providing such resources that is the country of origin of such resources or a Party that has acquired the genetic resources in accordance with the Convention. Such sharing shall be upon mutually agreed terms.

Similarly, Articles 6 and 7 of the Protocol provide that the access to genetic resources and traditional knowledge should be regulated by each party, and that it should occur on the basis of prior informed consent (PIC).[16] PIC is a doctrine that is important to use in any context in which private bargaining is taking place over social values. For example, in the original context in which PIC was used (acceptance of hazardous waste shipments), it makes a lot of sense to create a structure whereby the state is informed about the transactions

Rights—but very little effort has been made to resolve the problem of unrecognized rights in useful biological resources more generally.

[15] On the private value of biodiversity in bioprospecting, see Simpson et al. (1996) and Rausser and Small (2000). On the social value of biodiversity see Goeschl and Swanson (2002) and Sarr et al. (2008). These values are likely to significantly outweigh private values.

[16] The doctrine of 'prior informed consent' was first developed in the context of the Basel Convention on Transboundary Shipments of Hazardous Wastes, and provides the basis for bargained-over solutions in an environment of complete and shared information.

being undertaken by any private agents capable of having a substantial impact on social outcomes. It simply provides the mechanism by which a state is informed about such private negotiations, and is given final authority over the conclusion of such private negotiations (and the information with which to undertake its own decision-making process).

The difficulty with establishing a well-informed bargaining environment within which negotiations are to occur is that there is nothing as yet over which to bargain. As described in the preceding section, the basis for bargaining would have to be a recognized right to the information emanating from natural genetic resources (*sans* improvement) and this does not yet exist. Informed bargaining is important once the foundations for bargaining are already in place. This is not yet the case with regard to the informational values of genetic resources.

Another tack is taken as regards the biodiversity bargaining problem in Article 10 of the Nagoya Protocol. This Article is entitled 'A Global Multilateral Benefit Sharing Mechanism', and provides as follows:

> Parties shall consider the need for and modalities of a global multilateral benefit-sharing mechanism to address the fair and equitable sharing of benefits derived from the utilization of genetic resources and traditional knowledge associated with genetic resources that occur in transboundary situations or for which it is not possible to grant or obtain prior informed consent. The benefits shared by users of genetic resources and traditional knowledge associated with genetic resources through this mechanism shall be used to support the conservation of biological diversity and the sustainable use of its components globally.

No concrete details emerged on the mechanism. The creation of a benefit-sharing mechanism is one issue that has been kicked into the long grass of biodiversity bargaining. The intimation in Article 10 is that we should await the establishment of a Protocol to the Protocol for the establishment of this fund.

In short, the Nagoya Protocol has yet to add anything of substance to the previous solution concepts under the CBD. The fundamental problem of using private bargaining as a resolution concept lies in the absence of internationally recognized rights in the informational values flowing from unmodified genetic resources. The Nagoya Protocol has created a more formal structure for providing access to such resources, but it has done nothing to address the fundamental property right failure that lies at the base of this problem.

11.4.6 Outside the box? The use and usefulness of REDD

One of the more substantial efforts to deal with the creation of a mechanism for managing deforestation and land conversion remains 'outside the box'—i.e. it is the programme for the reduction of emissions from deforestation and land degradation (REDD). REDD had its initiation in the Bali Declaration at the United Nations Framework Convention on Climate Change (UNFCCC) conference there of 2007 (COP13). At that meeting a roadmap was agreed for the adoption of a Bali Action Plan for compensating forested countries for activities designed to prevent their deforestation or degradation. The Copenhagen Accord of December 2009 adopted at COP15 then incorporated the recognition of a responsibility of developed countries to compensate developing countries for the avoidance of deforestation and degradation. A formal resolution was then adopted at COP16 providing for the establishment of *avoided deforestation* as one of many acceptable mitigation strategies under the UNFCCC. This constituted the formal initiation of the so-called REDD + programme of mitigation measures.

This has resulted in a plethora of international programmes targeted at the creation of mechanisms for transferring funding from developed to developing countries, in return for credits to be usable under an emissions restriction programme under the UNFCCC. Much of this activity is still within the pilot phase of these programmes, but the basic outline of transferring funds in exchange for carbon credits is clear. The precise mechanism for ascertaining baselines, or determining the level of credit achieved, remains to be determined; however, the idea of paying for non-deforestation is becoming entrenched via these REDD programmes.[17] It is stated on the UN REDD + site that it is hoped that US$30 billion should be transferred annually from developed to developing country parties under the auspices of the REDD + non-deforestation programmes.

REDD + is a programme that has developed out of a very different set of motivations for the prevention of deforestation, relative to the biodiversity regulation problem. It is a programme based on the observation that approximately one-quarter of all carbon emissions result from deforestation rather than fossil fuel consumption. This means that it is critical for any solution to the climate change problem to incorporate some means for regulating land use as well as fossil fuel use, in order to control carbon releases.

The primary problem with REDD as a biodiversity regulation mechanism is that it is an instrument that is targeting a related but not perfectly correlated objective—i.e. the sequestration of carbon in the biosphere. There are many examples of carbon sequestration schemes that would in fact be entirely destructive of biodiversity goals while advancing carbon sequestration (e.g. seeding of oceans). There are even examples of schemes that would advance forestation while diminishing diversity (e.g. mono-cultural plantation forestry). These are extreme examples, but illustrative of the fact that the two goals do not necessarily go hand in hand.

[17] Examples of the facilitators of various REDD programmes include UN REDD; GEF; Norwegian Forestry Plan.

In general, all policy economists know that it is best to have as many instruments as there are objectives being pursued. If the goal is to pursue both maximum carbon sequestration in the biosphere *and* maximum biodiversity, then the best way to do so is to have an instrument targeting each individually.

Of course we live in the world of the second-best, and so the real question for consideration is whether REDD + is a mechanism that might potentially afford the needed mechanism for doing deals in non-conversion. That is, could the problem of a global biodiversity regulation mechanism be shoe-horned on top of this mechanism created for the purposes of carbon sequestration?

The challenge is identifying the correct bargaining frontier of the problem that is being confronted. States that are attempting to purchase the development rights of others with regard to fossil fuel-based development are purchasing one thing. States that are attempting to purchase the development rights of others with regard to land conversion are purchasing another. We would argue that both the bargaining frontiers exist and are distinct from one another. The distributional problems to be resolved are two—one concerns the value of the life sciences industries and the other concerns the value of fossil fuel-based industries. There is a natural inclination to want to combine the two problems—since the purchase of non-conversion rights is one possible solution concept to both—but this both conflates two distinct bargaining frontiers and unnecessarily narrows the range of potential solution concepts for carbon sequestration.

In short, if the problem is biodiversity, then it makes sense to both fashion its own instrument and to face its own bargaining frontier. REDD appears to be an attempt to hit two targets with a single payment—i.e. to purchase two objectives at the price of a single transfer. There may be a very small set of lands where the optimal use is both carbon sequestration and biodiversity conservation, but in general it is likely that the two goals will lead towards very different targets.[18]

11.5 REFRAMING THE GAME: RATIONAL THREATS AS A RESPONSE TO UNFAIR BARGAINING

11.5.1 Strategic destruction as a rational threat

In terms of the bargaining framework, the IC contract of the CBD leaves the South indifferent between cooperation and non-cooperation.[19] It is an

[18] Collins et al. (2011) show that the coincidence of biodiversity (charismatic species and pleiotropic—dependent on habitat—species) and carbon rich forests is far from uniform.

[19] Inequitable distributions such as this are frequently at the bottom of non-cooperation, as in the case of the Pacific Salmon Treaty, the European Sulphur Protocols of the late 1980s, and so on (Miller et al., 2000; Mason and Swanson, 2003).

extreme point contract. How problematic is this? After all no one is worse off. The problem is that the ability to shift the conflict point by one or other party confers the ability to 're-frame' the bargaining game to their own advantage. Shifting the conflict point, or threatening to do so, can be a rational bargaining strategy and confers another form of bargaining power.[20]

Suppose in Figure 11.2 the South now threatens to push the conflict point from U^a to U^D, reducing the North's conflict payoff without affecting its own. Reframed in this way, the Nash solution now becomes (U_N^T, U_S^T), which confers a greater share of the global surplus to the South.

At first glance it would appear that the asymmetric endowments would result in equivalent and reciprocal threat capacities: the 'technology-rich' North could threaten to reduce R&D, while the 'gene-rich' South could threaten to limit the supply of reserves, resulting in no real bargaining advantage.[21] However, this ignores the question of credibility. One obvious means of making a credible commitment is for the party concerned to threaten destruction of the required assets, should the parties fail to reach agreement on the basis of cooperation. Here there is a clear asymmetry in bargaining capacities: the South can credibly threaten irreversible destruction of its environmental resources, but the North cannot credibly threaten to destroy human capital or information. Furthermore, the assumption of irreversibility means this threat contains a 'natural' commitment mechanism. The asymmetry in capital endowments means that only the South can satisfy the necessary conditions for a credible threat in this bargain.

While the destruction of resources as a bargaining ploy sounds alarming, it has been noted in other contexts as a ploy to secure bargaining power.[22] Furthermore, Gatti et al. (2011) demonstrate that a solution such as U^T associated with the destruction conflict point U^D is possible in the agricultural biotechnology case, which confers a larger share of the surplus to the South. Another implication is that, since the South will be no worse off under conflict

[20] The strategic use of rational threats first analysed by Nash (1953) has been extended in several directions. More recent work in cooperative game theory has focused on dynamic games with inefficient outcomes, such as strategic destruction of the bargaining surplus. Busch et al. (1998) build on work by Shaked and Sutton (1984) to examine the equilibrium strategies giving rise to an asymmetric case in which one party has the capability of credibly destroying the cooperative surplus. Our case mirrors that of Busch et al. (1998).

[21] Parallels can be easily drawn between this type of threat for the North and the trade restrictions and limitations on technology transfer that have been the focus of the strategic trade literature (e.g. Krugman, 1979; Lai and Qiu, 2003).

[22] For example, Karp (1996) provides a theoretical analysis of the incentives for strategic destruction by a monopolist producing a durable good. This draws from a wider literature in industrial organization. Stranlund (1999) discusses an analogous case in which the bargaining outcome is influenced by strategic sunk investments.

post-destruction, destruction might actually be undertaken rather than simply threatened.[23]

From the perspective of a bargaining problem, strategic destruction can be a rational response to an inequitable bargaining solution like the IC contract of the CBD. Each party brings asymmetric yet complementary inputs to the negotiating table. This in itself suggests an equitable resolution to the bargaining problem. An inequitable outcome would leave one party, the North, vulnerable to strategic destruction in the South.

11.5.2 Strategic threats in practice

Such threats have been witnessed at local as well as international levels. In Latin America farmers who were offered an IC contract retorted 'Bueno, corto todo' (OK, I'll cut it all) when no compensation was offered for the existing stock of forest resources (World Bank, 2003). But there are several cases of international negotiations on biodiversity that can be interpreted from the perspective of bargaining with strategic threats.

The best documented examples concern the governments of Cameroon, Ecuador, and Guyana. In Cameroon in 2008 the Minister of Forestry, Joseph Thatta, made a clear statement of what the government perceived to be a fair share of the cooperative surplus, while effectively redefining the conflict point in the negotiations with international conservation organizations over the Ngoyla-Mintom forest. An annual fee of US$1.6 million for 830,000 hectares of biodiverse tropical forest was requested to prevent the concessions being sold to logging companies (see *The Economist*, 2008). Rough calculations suggest that the global value in terms of carbon sequestration alone is double the value of the logging concessions, so conservation is on the bargaining frontier.[24] In the absence of any offers, in March 2009 the government made good on its threat and the process of determining forest concessions began. In terms of the bargaining framework, at this point the process appeared to be stuck at the conflict point with the bargaining frontier contracting.

[23] Busch et al. (1998) show that this kind of outcome can be an equilibrium strategy in a sequential game with trigger strategies. That is, the simple solution presented here is not specific to the one-period analysis but remains possible in a dynamic context.

[24] The 830,000 hectares of forest in the Ngoyla-Mintom store over 200 million tonnes of carbon dioxide (assuming a conservative 250 tonnes of carbon dioxide/ha). Assuming that conservation reverses the 1 per cent trend in deforestation, and assuming emissions of 160 tonnes of carbon dioxide/ha from logging, at US$3/tonne of carbon dioxide, payments for carbon through the REDD scheme would generate credits with a net present value of US$64 million (over 30 years at 5 per cent discount). This exceeds the US$26 million in logging concession fees (*The Economist*, 2008).

The response to this apparent impasse has been noteworthy. The authors' conversations with institutions such as Conservation International point to the impasse resulting from a coordination problem between conservation organizations, rather than the absence of demand for such conservation projects. A number of attempts at coordinating conservation interests have arisen in response, and several conservation project proposals, including carbon finance, have been presented to the Ministry of Forestry in Cameroon.[25]

Several developments have taken place since that time, and in July 2012 the World Bank and the Global Environment Facility issued a press release in which they announced an extension of their decade-long involvement in the forest sector, and a development grant of \$3.5 million (\$ in 2012) for sustainable development of the Ngoyla-Mintom forest area. Part of the motivation for this involvement was to ensure conservation prior to the construction of new roads and the completion of already functioning roads in the forest area (World Bank, 2012). The amount of money being proposed is, by the admission of the project document itself, rather small. It appears that the bargaining stance of the Ministry of Forestry has succeeded in focusing ideas among donors on the importance of forests in Cameroon and the need for initiatives that transfer more than the incremental cost of conservation and which have long-term impacts on development.[26]

Similar threats were issued by President Rafael Correa of Ecuador in relation to the Yasuni–Ishpingo Tambococha Tiputini (Yasuni–ITT) region of Ecuador, which lies in the Amazon rainforest, at a meeting of the United Nations in September of 2007. Again, the conflict point and the share of the surplus were clearly defined, albeit under different circumstances to those of Cameroon. The conflict point was defined as the development of the oil fields beneath the Yasuni–ITT region. The share of the cooperative surplus, arising from leaving oil in the ground, would include compensation for lost oil revenues from the international community, which resembles the incremental cost component, and carbon credits amounting to the foregone carbon emissions, reflecting a payment for the stock of carbon. This contractual solution, which would conserve 38 per cent of Ecuador's land from damage by extractive industries, bears more than a passing resemblance to the optimal contract under strategic threats defined by Gatti et al. (2011), which confers a payment for the potentially destructible stock of resources rather than simply the incremental change under conservation. With funds to be administered by the UN and held in the Yasuni–ITT trust, and conditional on continued conservation, this initiative has been more successful than in the case of

[25] See, for example, the Ngoyla Mintom campaign: <www.NgoylaMintom.blogspot.com>; <http://dl.dropbox.com/u/10799684/NgoylaMintom2E.pdf>; <http://www.thegef.org/gef/content/cameroon-gef-grant-achieve-sustainable-development-local-communities-ngoyla-mintom-forest>.

[26] <http://web.worldbank.org/external/projects/main>.

Cameroon. In the first instance numerous pledges of finance were received, and in April 2010 a deal worth $3 billion was signed between the Ecuadorean government and overseas governments to support the initiative. After receiving pledges totalling more than its goal of $100 million by its deadline, the Ecuadorean government finally announced in early 2012 that it would move forward with the Yasuni–ITT Initiative.[27] It can be argued that this is one case in which strategic bargaining changed the nature of the solution.

In the months prior to this, in November 2009, a significant bilateral forest conservation agreement was signed between the governments of Norway and Guyana to conserve 50 million hectares for an investment of £150 million. Previous offers had been made by Guyana's president Bharrat Jagdeo to the UK government in 2007 to conserve rainforests in return for development aid and technical assistance, but to no avail. The agreement with Norway arose only after the timber value of the forest and a potential development plan were revealed to the international community. President Jagdeo's 'show me the money' approach has been described as a threat or even 'blackmail' in some quarters. What this chapter reveals is that such actions reflect a credible bargaining position based on asymmetric endowments.[28]

These examples represent attempts to dislodge the status quo, and certainly represent active use of threats, or at the very least, a laying bare of the structure of the bargaining game. Threats are not the only responses to the status quo that have been witnessed in the realm of biodiversity. The formation of the Group of Like-Minded Megadiverse Countries (LMMC) represents an alternative means by which to garner bargaining power, dislodge current solutions, and improve benefit sharing. In the context of the bargaining problem discussed here, this could represent an attempt to develop a credible threat, or an attempt to influence the sharing rule. In sum, these recent responses support the main finding here: that current solutions are unlikely to be long-lasting despite ostensibly solving the externality problem for now.

11.6 CONCLUSION

We have several conclusions to report on the problem of and policies for regulating global biodiversity.

[27] <http://www.sosyasuni.org/en/>.

[28] It is possible that the short history of Norway's involvement in Indonesia could be viewed in a similar way: see <http://www.redd-monitor.org/2012/05/25/after-one-year-indonesias-forest-moratorium-isnt-working/>, or <http://www.nytimes.com/2010/11/29/world/asia/29iht-indo.html>, although the issues here seem to concern monitoring more than strategic destruction and bargaining.

First, it is important to recognize that the problem of global biodiversity regulation is distinct from many of the smaller externality-driven policies regarding land-use management and conservation. These are local, regional, and national biodiversity policies addressed to the internalization of the broader values of unconverted lands. There is a function to be served by reason of sharing information widely on cost-effective local policies, but this has nothing to do with global biodiversity regulation—a different problem.

Second, the problem of global biodiversity regulation has foundered over the past twenty years. There have been a few attempts at creating policies for conservation under the broad rubric of the CBD: principally incremental cost contracts and benefit-sharing regimes. We have argued here that the former represents an attempt to place the providers on their participation constraint, while the former has accomplished nothing at all to date.

Third, the most promising regime for land conversion at the global level exists at present under the climate change regime. REDD + provides a basic mechanism for making transfers to developing countries in exchange for carbon credits, and it has been ushered in rapidly to great fanfare. The problem with using a carbon sequestration mechanism for regulating land conversion is that these are two distinct problems. At a minimum there is the argument that two policy objectives warrant two distinct instruments. At worst, there is the possibility that the biodiversity problem is being subsumed into the climate change problem—i.e. it is assumed that it is solved when the climate change land-use problem is addressed. Nevertheless, these are theoretical issues at present, and it is interesting to note the recent developments under the UNFCCC regarding deforestation issues, and to ponder why the major efforts at global land-use regulation have occurred in the context of a climate regime (rather than the biodiversity one).

Fourth, we have described in passing the manner in which a global land-use policy mechanism should operate. A transfer mechanism needs to be put in place that enables payments to those countries in correspondence and for each year in which they do not convert areas of existing natural habitats. This implies a long time horizon of ongoing payments for unconverted lands, but this is precisely the sort of mechanism which the biodiversity bargaining problem describes as its solution. There needs to be some means of sharing the benefits of land-based development between those states that have converted with those that have not.

It is important to begin thinking more generally about the great environment and development conventions as questions of cooperation over the production of joint surplus from such industries. The climate change regime should probably be thought of as a problem of deciding how to distribute the gains from fossil fuel-based development between those who have had it and those who never will. Similarly, the *global* problem of biodiversity regulation has little to do with internalizing local or regional externalities (such as watersheds) or with conserving amenities (such as elephants). The international policy problem of regulating global biodiversity concerns the determination of the total converted land area that will provide the optimal ratio of inputs to and outputs from biological industries. Again the fundamental problem at its heart concerns determining the distribution of gains between those states that have previously developed their lands, and those that agree never to do so. The realization of real policies on global biodiversity regulation awaits the recognition of these fundamental bargains that must be made. Until then, our analysis (and the current record) demonstrates that we can expect to see continuing conversion and deforestation in those countries that are going uncompensated—according to their perceptions of fairness.

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Part V

Policy Instruments and Incentives

12

Do Biodiversity Policies Work? The Case for *Conservation Evaluation 2.0*

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12.1 INTRODUCTION

Biodiversity loss results from the destruction and degradation of habitats, the harvesting of plants and animals for human use, competition from invasive species, and climate change (Slingenberg et al., 2009; Barnosky et al., 2011). Despite decades of investments to slow or reverse this loss, we lack evidence on whether and under what conditions conservation actions can be effective (Ferraro and Pattanayak, 2006). This chapter reviews the most recent evidence on the performance of commonly used conservation actions and identifies gaps in what we know about their effectiveness. As most threatened species and habitats are found in tropical developing countries (Hoffmann et al., 2010; Myers et al., 2000), we focus our analysis on the three most commonly employed actions in these countries: protected areas (PAs), payments for ecosystem services (PES), and decentralization of natural resource management.[1,2]

[1] We focus on ecosystem structure and function as a proxy for biodiversity. This focus is consistent with the working definition used by the Convention on Biological Diversity (CBD). Because the scope of the original definition of biodiversity was so broad, and because of the high correlation between the number of species, habitat quality, and quantity and other measures of biodiversity, the CBD has endorsed the ecosystems approach for the implementation and evaluation of conservation policies (CBD, Slingenberg et al., 2009).

[2] Forest decentralization is not a single well-defined policy. The literature has pointed out the multiple connotations of the term (e.g. Larson and Soto (2008) and Larson (2002) discuss in detail the multiple definitions; Coleman and Fleischman (2011) discuss the differences in the nature of the forest decentralization measures in Kenya, Uganda, Bolivia, and Mexico). Our emphasis in this chapter is on quantifying the impact of changes of the management authority on terrestrial ecosystems. For this reason, we use 'decentralization' as a term reflecting the redistribution of management authority from a higher to a lower level (communities or local governments).

Of course, these three approaches are also used in developed countries, and many of our conclusions apply equally there. We also briefly overview other common measures, such as integrated conservation and development projects and forest certification schemes, and find very little empirical evidence on their effectiveness.

PAs, like national parks and reserves, are the most commonly used tool for biodiversity conservation in developing countries: about 15 per cent of tropical developing countries overall is formally protected (World Database on Protected Areas 2011, available at <www.wdpa.org>). PAs place legal restrictions on human access and use within their boundaries. In contrast, PES schemes are more recent and seem to be concentrated predominantly in Latin America and China.[3] Unlike PAs, which rely on negative incentives to induce behavioural change, PES use positive incentives in the form of conditional payments to landowners in return for not converting habitat of high conservation value to other uses (Pattanayak et al., 2010). Decentralization approaches also rely on positive incentives, but they do so indirectly by decentralizing management authority to local actors (e.g. municipalities, communities). This decentralization is believed to create local incentives for sustainable natural resources by making it more likely that those who bear the costs of conservation also reap the benefits (Larson, 2002). Decentralization is also believed to enhance conservation outcomes because local actors may be better monitors of natural resource regulations and hold local governments accountable to otherwise marginalized groups (Larson, 2002; Larson and Soto, 2008; Coleman and Fleischman, 2011).[4]

Although theory from economics and political science suggests that all three conservation approaches can be effective under certain assumptions, the approaches often fail in practice.

For example, the theory underlying all three approaches assumes the existence and effectiveness of institutions and the rule of law (Hayes and Ostrom, 2005; Larson and Soto, 2008). Yet, developing countries are plagued by uncertain property rights, widespread corruption, and the absence of strong institutions that can effectively coordinate across scales to disseminate information, reduce transaction costs, and monitor and enforce laws (Heltberg, 2001; Vincent, 2010). Furthermore, the theory assumes that the three

[3] As of 2010 there are pilot PES programmes in Bolivia, Brazil, Colombia, Costa Rica, Ecuador, Mexico, Nicaragua, Venezuela, Kenya, South Africa, and China (Vincent, 2010). Jack et al. (2008) mention a PES-like scheme in Indonesia (Rewarding Upland Poor for Environmental Services (RUPES)).

[4] There is a very large literature (primarily comprised of case studies) that explores under what conditions decentralization can lead to the sustainable management of natural resources. See Larson and Soto (2008) for a review of recent studies. For theoretical arguments for the decentralized provision of local public goods, see Besley and Coate (2003). For research on the broader issues of the relationship between institutions and economic growth, refer to Acemoglu et al. (2004) and Besley and Persson (2011).

approaches are applied to important habitats that are threatened with conversion to other land uses. In reality, there is a strong selection bias. PAs are often targeted at lands with the least political resistance to their establishment, and thus typically face the least anthropogenic threat (Andam et al., 2008). Similarly, PES contract recipients volunteer their least profitable and, hence, threatened lands (Pattanayak et al., 2010; Ferraro et al., 2012). Decentralization is not immune to such bias either: communities that already have a record of good ecosystem management are the most likely recipients of increased management authority (Bowler et al., 2011).

The selection bias in the placement of interventions and the lack of effective institutions have cast doubt on the effectiveness of conservation investments and have spurred numerous calls for rigorous empirical evaluation of conservation policies and programmes (Kleiman et al., 2000; Pullin and Knight, 2001; Salafsky et al., 2002; Salafsky and Margoluis, 2003; Sutherland et al., 2004; Saterson et al., 2004; Frondel and Schmidt, 2005; Ferraro and Pattanayak, 2006; Carpenter et al., 2009; Pattanayak et al., 2010). These calls have highlighted the need for policy to be grounded in a firm understanding of *whether*, *under what conditions*, and *how* conservation programmes work. Translating such knowledge into policy can improve the performance, cost-effectiveness, and sustainability of conservation investments.

In section 12.2, we briefly describe quasi-experimental study designs, which can isolate the causal impacts of conservation interventions when they are not randomly assigned, and contrast these with common designs in the conservation literature. Then we summarize the current evidence for the effectiveness of PAs, PES, and decentralization, and briefly examine other common conservation approaches. Section 12.3 identifies major trends in the evidence and highlights the main lessons for biodiversity conservation. Section 12.4 calls for a new programme of research—*Conservation Evaluation 2.0*—that uses better theory, better methods, and better data to fill our knowledge gaps about what works and what does not in protecting biodiversity.

12.2 WHAT HAS WORKED AND WHAT HASN'T?

12.2.1 Empirical designs and methods

When natural scientists empirically assess the performance of a conservation action, they often employ one of two designs: (a) they compare outcomes (e.g. deforestation, poverty) in areas with and without exposure to the conservation action, or (b) they compare outcomes before and after the conservation action is implemented. 'With–without' analyses implicitly assume that (1) the areas with and without the conservation action are similar in terms of their expected outcomes in the absence of the conservation action (i.e. similar in characteristics that affect outcomes, such as accessibility, suitability for agriculture, and proximity to markets); and (2) there are no spillover effects from the conservation action to 'unexposed' areas. 'Before–after' analyses assume that the level of the outcome (or its trend) before a policy is enacted would remain constant after the policy is enacted (Nagendra, 2008).

If any of these assumptions fail, the estimates of conservation effectiveness will be biased (Ferraro and Pattanayak, 2006; Ferraro, 2009; Joppa and Pfaff, 2010).[5] Consider the case of PAs in tropical forests. First, deforestation rates may change after the establishment of PAs for reasons other than protection (e.g. commodity prices), thus invalidating a simple before–after comparison of deforestation (Nagendra, 2008; Joppa and Pfaff, 2010). In other words, without a comparison with an unprotected control group, changes in deforestation rates within the protected area cannot be unambiguously ascribed to protection, but could be the result of unrelated factors. Second, PAs, like other conservation interventions, are not placed randomly across the landscape. Instead, they tend to be established among poor communities on lands far away from cities and unsuitable for agriculture or urbanization (Pfaff et al., 2009; Joppa and Pfaff, 2009, 2010; Andam et al., 2010). These lands may experience below-average deforestation even in the absence of protection. In such cases, simple contrasts of areas with and without protection will yield erroneously high estimates of the PA effectiveness: even in the absence of protection, deforestation rates on the protected lands would be lower than the average deforestation rates of unprotected lands. Third, the establishment of a protected area may displace the extractive activities to nearby buffer zones (Armsworth et al., 2006; Joppa and Pfaff, 2010). In this case, if we compare deforestation within the park with deforestation in the buffer zones, the estimate of the impact of protection will be biased upwards because deforestation rates in the unprotected areas would have been lower had the protected areas not been established.

Similar violations of key assumptions arise for PES and decentralization. Participation in PES programmes is often driven by the lack of profitable alternative uses for the land (Pattanayak et al., 2010). For example, large forested tracts owned by absentee landlords and with steeper slopes (i.e. low agricultural suitability) have a higher probability of enrolment in a PES programme in Costa Rica (Arriagada et al., 2009; 2012). Likewise, decentralization in some areas occurs on forested land that is already well-managed locally and thus would be in above-average condition even in the absence of

[5] For example, comparing results obtained from methods that do not control for selection bias or outcomes changes over time to methods that do control for them, Andam et al. (2008) found that methods that fail to control for them overestimate the effectiveness of protected areas in Costa Rica by more than 65 per cent.

decentralization (Ferraro et al., 2012). For example, effective local governance is often attributed to high levels of social capital within communities, which may make these same communities better at monitoring forests and enforcing use rules in the absence of decentralization (Baland et al., 2010). In contrast, other communities receive the authority to manage their forests because previous forest degradation renders the forests less valuable to authorities (Baland et al., 2010). These lands are more likely to be in below-average condition in the absence of decentralization (i.e. with–without estimators would yield erroneously low estimates of impacts). Ultimately one cannot estimate the effect of a conservation programme without understanding the programme assignment process: why are some areas exposed to the programme while other areas are not?

Understanding programme assignment is critical because all credible impact evaluations must estimate what would have happened in areas exposed to the programme if they had not been exposed—i.e. the *counterfactual* outcomes in the absence of the programme. The average causal effect of a programme is the difference between the average observed outcome with the programme and the average counterfactual outcome without the programme. Using the outcome prior to the programme as an estimate of this counterfactual outcome is problematic if conditions have changed, as they almost invariably will have. Estimating the counterfactual outcome by using the outcome of a control group unaffected by the programme is problematic when conservation actions are not randomly assigned, but rather deliberately assigned for reasons that are correlated with the outcomes themselves (e.g. good stewards are more likely to have management authority devolved to them than poor stewards). These reasons that affect both the conservation outcomes and when and where the conservation action takes place can mask or mimic the impacts of the conservation action: they are confounding variables whose effects must be 'blocked' in order to identify the causal effect of the conservation programme. In the statistical terms of a regression equation, the non-random assignment of conservation programmes translates into a correlation between the policy variable (the treatment) and the error term, which biases the estimated coefficient of the policy variable. The direction of the bias depends on the sign and the magnitude of the correlation between the policy and the error term (Greenstone and Gayer, 2009).

To mitigate the bias, experimental and quasi-experimental designs from the programme evaluation literature can be used. Given that experimental designs are absent from the biodiversity conservation policy literature (Ferraro, 2009; Greenstone and Gayer, 2009; Joppa and Pfaff, 2010),[6] we focus on three

[6] We are aware of only one proposed study whose design employs a group randomized control trial in which the payments for forest protection are randomly assigned to some villages and not to others: UNEP, National Environment Management Authority (NEMA) Uganda;

commonly used quasi-experimental designs: matching, instrumental variables, and difference-in-differences (DID) designs (Pattanayak, 2009). Matching designs reduce bias by matching units (e.g. forests, farms, communities) affected by the conservation policy, called treated units, with observationally similar units that are not affected by the policy, called control units (Imbens and Wooldridge, 2009). Matching assumes that once we have controlled for, or conditioned on, key observable characteristics that affect the outcome and exposure to the conservation programme, the programme can be assumed to have been 'as if' randomly assigned. In other words, there are no systematic unobservable differences between treated and 'observably similar' control units that could explain the presence or absence of a correlation between the programme and an outcome. In contrast, an instrumental variable design reduces bias by exploiting a variable that affects the programme assignment, but does not affect the outcome. This instrumental variable creates a natural experiment for some sub-population of the affected units: for these units, their exposure to the programme is for reasons unrelated to their potential outcomes with and without the programme, just like one would observe in a true experimental design (e.g. their potential deforestation with and without formal protection). For example, if PAs are more likely to be assigned where endemic mammals are present, but the presence of endemic mammals only affects deforestation rates through its effect on the likelihood of a parcel's protection, then the presence of endemic mammals can be used as an 'instrument' to identify a causal effect of PAs on deforestation. In practice, it is often hard to find instruments that are both strong (correlated with the intervention) and valid (do not affect the outcome or are affected by unobservable factors that affect the outcome). The third popular empirical design, the DID design, uses the difference in the before–after changes in the outcomes for protected and unprotected areas to estimate the causal effect of the programme (called the Before–After–Control–Impact, or BACI, estimator in ecology). The DID design assumes that the average trend of the control units represents the average trend of the treated units in the absence of treatment (perhaps conditional on some observable characteristics). In other words, any unobserved differences (i.e. systematic biases) between the treated and control units are additive and do not vary over time, and can hence be removed by taking the difference in the outcomes before and after the policy.

The three designs can be used independently or in combination. They can also be used with other common statistical approaches to measuring causal effects such as panel data designs. Panel data designs are essentially an

International Institute for Environment and Development (IIED) 2010. 'Developing an Experimental Methodology for Testing the Effectiveness of Payments for Ecosystem Services to Enhance Conservation in Productive Landscapes in Uganda', Proposal to the Global Environment Facility, Washington DC (available at <http://www.thegef.org/gef/node/2772>).

extension of the DID design and use repeated outcome measures before and after the programme starts to control for time trends in the outcomes. Although these designs hold great promise for estimating the effects of conservation actions, they can be challenging to implement (see Ferraro and Pattanayak, 2006, for a discussion). For example, none of these designs is immune to bias from spillovers from treated to control units, which may be common in conservation programmes. For example, protected areas may displace agricultural pressures (leakage) to neighbouring control areas, or they may increase enforcement of existing land-use laws in neighbouring control areas. If such spillovers are likely, one should either seek a control group that is unaffected by spillovers, or explicitly measure the spillover effects.

12.2.2 Empirical evidence on conservation policy performance

Protected areas

Table 12.1 summarizes the studies that use rigorous empirical designs to quantify the impacts of protected areas. These studies have focused predominantly on the effectiveness of PAs in preventing deforestation, most often measured as a binary outcome at the pixel level.[7] The results suggest that PAs are effective at reducing deforestation (e.g. Andam et al., 2008; Gaveau et al., 2009; Pfaff et al., 2009; Sims, 2010; Joppa and Pfaff, 2010; Ferraro and Hanauer, 2011), encouraged regrowth on previously cleared lands (Andam et al., 2013), had mostly negligible spillover effects (Andam et al., 2008; Sims, 2010), and reduced the incidence of forest fires (Nelson and Chomitz, 2011). Nevertheless, the estimated effects are much smaller than conventional before–after and with–without designs would imply.

A few studies have suggested that the impacts of the PAs are heterogeneous: they vary through time and in space according to the baseline characteristics of the area. For example, Ferraro et al. (2011) find that in Costa Rica the impact is greatest on land that has lower slopes, poor population, and is closer to major cities. They also find that in Thailand the impact of PAs on preventing deforestation is highest on land with lower slopes, but far away from major cities. Andam et al. (2008) find larger impacts of older PAs compared to newer PAs. Pfaff et al. (2011) compare the impacts across federal and state-managed parks and find that the intervention is more successful under the former. Nelson and Chomitz (2011) find that PAs have a positive impact on reducing forest fires, with the magnitude of the effect varying by geographic location, type of PA (strictly protected vs. multi-use), and the proximity to cities.

[7] Gaveau et al. (2009), Sims (2010), and Honey-Roses et al. (2011) employ a continuous outcome variable (percentage deforestation).

Table 12.1. Protected area studies using rigorous empirical analysis

Study	Location	Unit of analysis	Sample size (protected/ unprotected)	PA type	Methods	Outcome
Andam et al. (2008)	Costa Rica	Pixel	2,711/10,371 2,022/4,724	IUCN I-VI*	Matching	11% reduction in deforestation
Ferraro and Hanauer (2011)	Costa Rica	Pixel	Same as in Andam et al. (2008)	IUCN I-VI*	Matching	Trade-offs b/w deforestation and poverty reduction
Ferarro et al. (2011)	Costa Rica	Pixel	Same as in Andam et al. (2008)	IUCN I-VI*	Matching, PLM	11% deforestation reduction; trade-offs b/w deforestation and poverty reduction
Pfaff et al. (2009)	Costa Rica	Pixel	4,229	IUCN I-II	Matching, regressions	1–2% deforestation reduction
Ferraro et al. (2011)	Thailand	Pixel	Same as in Sims (2010)	IUCN I-II*	Matching	15% deforestation reduction; trade-offs b/w deforestation and poverty reduction
Joppa and Pfaff (2010)	147 countries†	Pixel	5% of PA in each country/4× unprotected area	IUCN I-VI	Matching	Deforestation reduction in over 75% of the countries in the sample
Gaveau et al. (2009)	Sumatra and Siberut	Pixel	463/423	Conservation and hydrological PAs	Matching, regressions	24% deforestation reduction‡
Haruna (2010)	Panama	Pixel	9,467/27,559	IUCN I-II	Matching	12–16% deforestation reduction
			8,372/27,121			12–15% deforestation reduction
Sims (2010)	Thailand	Locality	20,565	IUCN I-II	IV	7–19% deforestation reduction
Schwarze and Jurhbandt (2010)	Indonesia	Pixel	10,418/13,888	Lore-Lindu National Park	Matching	9.4% deforestation reduction

Nelson and Chomitz (2011)	Tropical developing countries	Pixel	Varies	All PAs[§]	Matching, LOESS	Some reduction in forest fires, impacts vary by intervention, time period, and distance to major city
Gimenez (2012)	Madagascar	Pixel	8,083/87,379 for 1990–2000	All PAs	Matching, regressions	5.47% deforestation reduction 1990–2000
			12,250/89,471 for 2000–2005			1.52% deforestation reduction 2000–05
Nolte and Agrawal (2012)	Amazon (sites in Brazil, Peru, Bolivia)	Pixel	5% of protected, 5% unprotected areas	All PAs	Matching, LOESS	Some reduction in forest fires, some evidence that impacts vary by PA management effectiveness
Andam et al. (2013)	Costa Rica	Pixel	1219/14,594	IUCN I-VI*	Matching	13.5% increase in forest regrowth on previously deforested parcels. No statistically different impact of IUCN I-IV PAs compared with V-VI PAs

* Indigenous reserves and wetlands were excluded.

† All with >100 sq. km PAs.

‡ Using the same study area and time periods, Gaveau et al. (2012) compare the deforestation outcomes of protected pixels with unprotected pixels in the conversion and production zones. They find that the conservation intervention significantly reduced deforestation when compared with conversion zones, but not in comparison to production zones.

§ The study aggregates PAs into strict PAs, multi-use, and unknown, based on the IUCN categories.

Decentralization measures

Table 12.2 summarizes the studies that use rigorous empirical designs to quantify the causal impacts of decentralization programmes on environmental outcomes. These studies find that the placement of decentralization interventions is associated with factors that also affect the measured outcomes, thereby invalidating simple comparisons between decentralized and non-decentralized resources. For example, Somanathan et al. (2009) observe that state-controlled forest plots had more forest cover at the baseline, were located on slopes away from roads and villages, and had large nearby forest stocks and low population density. Baland et al. (2010) find that community forests were located closer to the villages, while Edmonds (2002) finds that the villages with decentralized forests had higher levels of electricity and piped water access, were close to a local market and forestry offices, and received more agricultural assistance. All of these factors affect environmental and social outcomes in the absence of any decentralization programme.

In contrast to the PA studies that use measures of deforestation or fire as an outcome, almost all of the decentralization studies use measures of forest degradation (proxied by the amount of fuelwood collected, density of the canopy cover, forest regeneration, and lopping). Overall, they find limited evidence that forest management decentralization policies reduced forest degradation. Somanathan et al. (2009) find higher crown cover in pine tree forests with decentralized control, but no impact in broad-leaved forests which are more heavily used by households and more likely to be degraded (Baland et al., 2010). In another part of India, Baland et al. (2010) found that decentralization reduced lopping, but had no effect on tree cover, age of the trees (proxied by the tree diameter at breast height (DBH)), or the presence of saplings. Coleman and Fleischman (2011) find that, on average, the African forests in their sample (parts of Uganda and Kenya) experienced a negative, albeit insignificant, impact from decentralization, while the forests in the Latin American countries (parts of Bolivia and Mexico) were positively affected (however, only for Mexico were the results statistically significant). Using multi-period panel data at the district level in Indonesia, Burgess et al. (2012) find that increasing the number of jurisdictions (a broad form of decentralization) increases deforestation and that the impact is strongest immediately before local elections.

Based on the studies summarized in Table 12.2, the impact of decentralization policies in terms of reducing forest degradation and deforestation seems context-specific; it varies in terms of the scope, benefits, and the rights transferred to local populations. This variability is not surprising given the ambiguous definition of 'decentralization': in our reviewed studies, it can refer to increasing the decision-making authority of lower-level bureaucrats or of local users (Larson, 2002; Larson and Soto, 2008). In other cases, it can be

Table 12.2. Decentralization studies using rigorous empirical analysis

Study	Location	Unit of analysis	Intervention	Sample	Method	Outcome
Burgess et al. (2012)	Indonesia	Pixel	# political jurisdictions	Large # pixels	Poisson model	Decentralization increased deforestation
Coleman and Fleischman (2011)	Bolivia	Forest user group	National vs. municipal institutions	11 treatment, 42 control groups	Probit, matching	(+) forest investments (+) perceived forest quality (not statistically significant)
Andersson and Gibson (2007)	Bolivia	Municipality	National vs. municipal institutions	30 observations, 2-period GIS data	IV	No effect of municipal institutions on total or permitted deforestation; (+) impact on illegal deforestation
Pfaff et al. (2011)	Brazil	Pixel	State vs. federal management	40,321 pixels	Matching	Federal PAs reduced deforestation, impact varies by type of PA
Edmonds (2002)	Nepal	Household	State vs. local	1200 households	Matching, IV, RD	14% reduction in fuelwood collection
Baland et al. (2010)	India	Forest transect	State vs. local	83 villages, 399 forest transects	OLS and Clogit w/ village FE	20–30% reduction lopping; no impact on DBH, canopy cover, #saplings, or fuelwood collection time
Somanathan et al. (2009)	India	Pixel	State vs. council managed forests	355 treatment, 582 controls for broad-leaved pixels; 318 treatment, 504 controls for pine trees	Regressions, Matching	Forest degradation (% crown cover): (+) impact for pine tree forests, no impact for broad-leaved forests, reduced cost of conservation
Heltberg (2001)	India	Village, household	Local institutions	180 households, 37 villages	IV	No impact on degradation (HH firewood dependence, state of the forest)

(*continued*)

Table 12.2. continued

Study	Location	Unit of analysis	Intervention	Sample	Method	Outcome
Bandyopadhyay and Shyamsundar (2004)	India	Household	Community management	8,307 households in 524 villages	Matching	Fuelwood consumption increase in villages with community management (some concerns with the model, though)
Coleman and Fleischman (2011)	Kenya	Forest user group	Community management	14 treatment, 57 control groups	Probit, matching	(–) forest investments (–) perceived forest quality (not statistically significant)
Coleman and Fleischman (2011)	Mexico	Forest user group	National vs. community management	19 treatment, 21 control groups	Probit, matching	(+) forest investments (+) perceived forest quality
Coleman and Fleischman (2011)	Uganda	Forest user group	National vs. community management	42 treatment, 102 control groups	Probit, matching	(+) forest investment* (–) perceived forest quality (not statistically significant)

Notes: Local = village or community level.
* Planting trees, seeds, and bushes.

associated with transfers of capital to local users or the establishment of property rights (Coleman and Fleischman, 2011). To better understand the impacts of decentralization, we need to clarify the mechanisms through which decentralization affects environmental outcomes. For example, Coleman and Fleischman (2011) propose accountability and empowerment as variables that moderate the effects of decentralization on forest quality and the welfare of local users.

Payments for ecosystem services

Table 12.3 summarizes the studies that use rigorous empirical designs to quantify the causal impacts of PES schemes. The studies tend to find reduced deforestation and increased reforestation taking place as a result of the participation in the PES schemes. None of the studies considers the impact on forest quality, as opposed to canopy cover (Pattanayak et al., 2010). All of the evidence comes from Latin American countries that have significantly more land under private ownership compared with the rest of the world (Vincent, 2010).

The effectiveness of the PES schemes depends on the programme design (e.g. where, to whom, and by whom the payments are made), the degree of compliance, and spatial spillovers (leakage) (Pattanayak et al., 2010). Previous studies have pointed out that the small impacts may be due to the poor initial targeting of PES schemes (especially in Costa Rica; Pfaff et al., 2008; Arriagada et al., 2012). Because participation in these schemes is voluntary, PES programmes are likely to suffer from moral hazard and adverse selection problems (Ferraro, 2008; Pattanayak et al., 2010; Ferraro et al., 2012). Thus attention to contract design and spatial allocation is crucial to achieve results in PES programmes.

Other conservation initiatives

Integrated conservation and development projects (ICDP) are widespread project-based interventions that aim to directly tackle the links between natural resource dependence, conservation, and poverty (Blom et al., 2010). Forest certification schemes provide financial stimuli for firms and farmers to adhere to defined environmental standards (Blackman and Rivera, 2010). Despite the long history and popularity of ICDP and forest certification schemes, we omit an extensive discussion of them because the number of rigorous impact studies is very small, with the evidence suggesting no impact from the interventions. For example, the only two studies that use rigorous empirical methods find no evidence that ICDP shifted households away from agriculture toward sustainable forest use in the Brazilian Amazon

Table 12.3. PES studies using rigorous empirical analysis

Study	Location	Unit of analysis	Sample	Methods	Outcome
Alix-Garcia et al. (2012)	Mexico	Farm plots	352 PSAH contracts, 462 controls	Matching and Tobit	10% deforestation reduction
Scullion et al. (2011)	Mexico	Farm plots	38 PES contracts, unspecified # controls	DID	34.8% deforestation reduction (pine/ oak forest)
					18.3% deforestation reduction (cloud forests)
Honey-Roses et al. (2011)	Mexico	Polygon*	425 treatment, 3,778 controls	Matching, DID	3–16% deforestation reduction in high-quality habitat
					0–2.5% deforestation reduction in lower-quality habitat
Sierra and Russman (2006)	Costa Rica	Farm plots	30 PES contracts, 30 controls	OLS	0.4 ha fallow −0.25 ha forests
Sills et al. (2008)	Costa Rica	Farm plots	44 PSA contracts, 119 controls	PSM and DID	3–10 ha natural forests
Arriagada et al. (2008)	Costa Rica	Tracts	1,019 PSA tracts, 519 controls	PSM and DID	25–35 ha reforested
Pfaff et al. (2008)	Costa Rica	Pixel	40 PSA pixels, 40–240 controls	PSM	<1% deforestation reduction
Arriagada et al. (2012)	Costa Rica	Farms	50 treated PSA farms, 152 control	Matching and regression	11–17% reforestation
Robalino et al. (2008)	Costa Rica	Pixel	925 PSA pixels, 925–4,625 controls	PSM	0.4% deforestation reduction

Note: * In contrast to pixels, these are of irregular shape and size. They result from the unique combinations of geospatial layer attributes and may therefore not coincide with farm plot boundaries.

(Weber et al., 2011; Bauch et al., 2012). Similarly, de Lima et al. (2008) find only small impacts of forest certification.

We have also omitted from the current discussion (1) conservation policies that are commonly used to target individual species, such as the US Endangered Species Act (Ferraro et al., 2007), Individual Transferable Quotas (ITQs), and measures associated with the Convention on International Trade in Endangered Species of the Wild Flora and Fauna (CITES); and (2) common conservation approaches in developed countries (e.g. PAs, easements). Furthermore, our discussion focuses on policies targeting the *symptoms* of unsustainable natural resource management (e.g. deforestation and forest degradation); we exclude from our analysis policies that might impact the underlying causes of biodiversity loss (e.g. international trade, macroeconomic policies, increasing demand for timber, food, and ranching) as the causal chain is longer and more complicated and the evidence even weaker. We discuss the implications for leakage and spillovers from conservation and related policies in section 12.4. Finally, although our emphasis in this chapter is on environmental outcomes, we note that there is an even greater paucity of evidence on the socioeconomic effects of conservation policies (e.g. Andam et al., 2010; Weber et al., 2011).

12.3 WHAT HAVE WE LEARNT SO FAR?

12.3.1 Protected areas seem to be effective

PAs seem to reduce deforestation consistently. A comparison of the effects of PAs and other interventions is possible for only one country. In Costa Rica, the effects of the PA system seem to be larger than the effects of the PES scheme (cost-effectiveness, however, is unknown). The larger effects may arise from multiple factors. For example, the PAs may have been established during periods of higher deforestation or may have existed for a longer period of time. Nothing is known about the effect of PAs on forest degradation except through the use of fire as its proxy. The evidence base on PES and decentralization in terms of deforestation and forest degradation is smaller and less consistent in its findings. Additionally, studies from the impact evaluation literature have noted the possibility of a large publication bias, with the majority of published articles skewed towards finding the expected statistically significant effect (Duflo, 2004; Greenstone and Gayer, 2009; Ravallion, 2009). To verify the representativeness of the reviewed PA studies, as well as the other studies described later in the chapter, it would be worthwhile to expand our review to catalogue and review unpublished working papers, theses, and reports to funding agencies.

12.3.2 Spillovers from conservation policies tend to be negligible

As already noted, conservation policies may result in changing the patterns of activities outside the targeted areas. Few studies have attempted to control for or measure spillovers. Some studies have tried to control for local spillovers by excluding from the control group areas that fall within a certain radius from the treated observations (Andam et al., 2008; Ferraro et al., 2011; Pfaff et al., 2011; Miteva et al., 2012*a*,*b*). Others have attempted to quantify the spillover effects directly by matching the unprotected areas near a protected area to areas unlikely to have been impacted by protection (Andam et al., 2008; Gaveau et al., 2009). Only one study has explicitly tested for the presence of spillovers in the context of PES: Alix-Garcia et al. (2012) find significant negative spillovers for the poorest quartile of their sample, and significant positive spillover effects for the wealthiest quartile. Overall, these studies find small positive or no statistically significant spillovers, possibly because the conservation impacts themselves are too small to generate spillovers.

12.3.3 Evidence limited to very few locations

Not only are studies with a credible empirical design rare, but the existing ones are not representative of the biodiversity 'hotspots'. For example, the majority of studies on PAs and PES focus on Costa Rica, which is an exceptional country in terms of development and biodiversity conservation. Very little evidence comes from other biodiversity-rich developing countries. Miteva et al. (2012*a*,*b*) provide evidence on the impacts of Indonesian PAs on deforestation, poverty, forest fires, species loss, and water quality. There are two global PA impact studies (Joppa and Pfaff, 2010; and Nelson and Chomitz, 2011). However, they use few controls for confounding covariates (necessarily because of the global scale of analysis), focus on a limited time period (2000–08), do not have true baselines for all the protected areas in the sample, consider a limited fraction of the country (only 5 per cent of the protected area in each country as in Joppa and Pfaff (2011)), or use approximations of the PAs where the exact borders are missing (Nelson and Chomitz, 2011). In contrast, the decentralization studies in Table 12.2 consider policies in East Asia (three countries), Latin America (three countries), and East Africa (two countries). However, almost all of these studies employ data collected from relatively small geographic areas and thus raise concerns about external validity.

12.3.4 No evidence on protecting ecosystem structure and function

The studies presented in Tables 12.1–12.3 consider the impact of biodiversity conservation policies on deforestation and forest degradation, which are *assumed* to be good proxies for species richness and ecosystem function. Moreover, the current literature does not consider where the conservation gains take place (with a few exceptions of studies looking at the heterogeneity of impacts according to the baseline characteristics of the area), what the resulting landscape configuration is (e.g. in terms of fragmentation and isolation of the forest patches), and whether the gains meet the threshold for the provision of certain ecosystem services (like improving water quality).[8] In other words, the degree to which deforestation and forest degradation can proxy for the ecosystem structure and function determines how useful these data are for telling us about the effectiveness of common biodiversity conservation approaches (Jack et al., 2008).

12.3.5 Impacts of conservation policies are heterogeneous

Baseline conditions

As Tables 12.1–12.3 suggest, the research focus has shifted away from quantifying the average impact of a policy to analysing the heterogeneity of policy performance as a function of the bio-physical and socioeconomic characteristics of the targeted areas. Nevertheless, such studies are few, and most compare the estimates for the average impacts within discrete groups of the data (exceptions include Ferraro et al., 2011, and Nelson and Chomitz, 2011).[9] The two studies that examine impact heterogeneity as a continuous function of the slope, distance to major cities, and poverty, find significantly non-linear impacts, with the PAs being most effective in areas with low baseline poverty, low slope (Ferraro et al., 2011), and closer to large cities (Ferraro et al., 2011; Nelson and Chomitz, 2011). In these studies, PAs had a negative impact on deforestation when the baseline poverty was high (Ferraro et al., 2011 in Costa Rica) or at intermediate distances to major cities (Ferraro et al., 2011 in Thailand). The only study considering impact heterogeneity in a PES scheme finds that the programme is more environmentally effective when the baseline poverty levels are low (Alix-Garcia et al., 2012).

[8] Recently, Sims (2011) has returned to her dataset of Thai PAs to examine if the PAs influence habitat fragmentation. She finds that PAs did prevent significant fragmentation overall, increasing average forest patch size by 20–33 per cent and forest patch density by 2–4 per cent. The more strictly protected wildlife sanctuaries appear to have encouraged consolidation of cleared patches and prevented forest fragmentation even in interior areas, consistent with core-focused enforcement patterns.

[9] Most of these studies do not allow us to assess whether there are statistically significant differences between the sub-groups. A notable exception is the study by Ferraro and Hanauer (2011), who use heteroskedasticity-robust variance adjustments to compute the confidence intervals (Ferraro et al., 2011, does something similar).

Type

In the case of PAs, Pfaff et al. (2011) find that federal parks are more successful at reducing deforestation in Brazil compared with state parks. Distinguishing between strictly protected and multi-use parks, Nelson and Chomitz (2011) find that the latter tend to result in reduced forest fire incidence in Latin America and Asia. Andam et al. (2013) find no difference in the amount of regrowth induced by strictly and less strictly protected areas. In the PES literature, case studies and descriptive approaches have suggested that the impact is likely to vary according to whether the government or users provide funds (Engel et al., 2008). However, the hypothesis that the effectiveness of PES schemes depends on the funding source has not been evaluated using rigorous quantitative approaches (Pattanayak et al., 2010). No clear patterns emerge for the different types of decentralization in Table 12.3.

Duration

Conservation policies often need time to effect changes (e.g. Baland et al., 2010; Jack et al., 2008). Some studies have circumvented this by focusing on the older policies. For example, Andam et al. (2008), Ferraro and Hanauer (2011), and Ferraro et al. (2011) consider separately the impacts of the PAs established before 1979 and after 1981; their results suggest that older PAs prevented more deforestation. In contrast, Nelson and Chomitz (2011) find a consistently larger impact of PAs on preventing forest fires when they restrict their treatment group from all PAs protected before 2000 to only those established between 1990 and 2000. Somanathan et al. (2009) focus on forests that have been decentralized for at least fifteen years. By discretizing the age of decentralized forest plots into older and newer groups, Baland et al. (2010) found that the impact of community-managed forests on lopping increases over time. None of the studies on PES has examined the heterogeneity of impacts through time. To our knowledge, no study has quantified how conservation effectiveness changes as a continuous function of time since protection.[10]

12.4 TOWARDS CONSERVATION EVALUATION 2.0

In this section, we draw on the literature in development impact evaluations and environmental and resource economics to highlight what the next generation of conservation impact studies should look like: we call this

[10] In order to do this, researchers either have to assume that selection does not change over time, or need to have panel data and an empirical design that allows selection to be a function of time. In other words, constructing the counterfactual is a substantial challenge in this case.

Conservation Evaluation 2.0. This new programme emphasizes better theory, better methods, and better data.

12.4.1 Better theory

Published empirical work in conservation policy science is typically disconnected from theories that describe how the interventions affect outcome. Elaborate theories help eliminate rival explanations for the observed empirical patterns, and thus help increase our confidence in the causal nature of our estimates (i.e. internal validity). Just as importantly, they help us move from understanding *whether* and *where* conservation programmes cause socioeconomic or environmental impacts to understanding *why* or *how* the programmes work (Ravallion, 2007, 2009; Ferraro et al., 2012). Understanding the heterogeneity of the programme effects and the mechanisms through which they propagate is crucial for interpreting the estimated impacts and determining the external validity (whether the impacts would be observed in other contexts), scalability, and expected persistence of the estimated impacts. We address each of these issues, as well as the way in which theory helps with assessing the internal validity of our studies, in more detail next.

Internal validity

The absence of a clear account of the mechanisms through which conservation programmes effect change, coupled with a limited understanding of the contexts in which they operate, raises concerns that there may be outside factors driving the post-programme outcome (Ferraro, 2009). For example, many protected areas are established to protect forests. In certain ecosystems, having more forests translates into a higher probability of natural forest fires and, frequently, into greater loss of tree cover, holding everything else constant. Unless we control for the probability of natural fires, we may conclude that protected areas are ineffective at protecting forests even though they may have substantially decreased the timber harvesting within their boundaries compared with counterfactual rates. Therefore, without theory to guide the model specification of quasi-experimental designs, we have to worry that omitted variables, inadequate controls for pre-treatment trends, and misspecification in the treatment selection model may mask the impact of the conservation intervention (Greenstone and Gayer, 2009).

Causal models with explicit assumptions (representing 'theories of change') can remedy this situation by identifying key confounding variables and appropriate samples with which to estimate counterfactual outcomes (Pattanayak et al., 2010). For example, theory suggests that local institutions and social capital are hard-to-measure often-omitted variables that are likely to bias programme evaluations (Deaton, 2010*a*). To control for such variables in an evaluation of decentralization performance, Baland et al. (2010) use a village-level block design and sample decentralized and non-decentralized forests within a village. Miteva et al. (2012*b*) use political economy theory to guide their choice of political factors (e.g. voting behaviour) that, in combination with bio-physical and socio-demographic variables, can help mitigate selection bias in their estimation of the causal impacts of Indonesia's PA on deforestation and poverty.

Better theory of change also helps identify the scale of the impacts from the conservation policy, and thus guides the analyst in the choice of the appropriate sampling frame. Valid empirical designs for estimating causal effects require that treatment and control units are independent. For this reason, observations from the control group that are likely to have been affected by spillover effects should be excluded from the control group. Yet, without a theoretical model of *whom* and *how* the conservation policy impacts, it is hard to know where exactly the spillover effects occur. Current attempts to deal with spillovers take an exploratory strategy by considering buffers of different width from the PA boundary (e.g. <2km, 2–4km, 4–6km, and 6–8km as in Andam et al., 2008). Yet, there is no theoretical justification for these buffers.

External validity

Under what conditions can we generalize the results to other contexts, given that evaluations of conservation programmes often use non-representative samples? Theory-based mechanisms that explain *how* and *why* the conservation intervention works (or doesn't) are necessary for out-of-sample predictions to forecast the impacts of conservation policies in new contexts (Heckman, 2010; Deaton, 2010*a*,*b*). The existing literature has taken an inductive strategy towards discovering the contexts that matter such as bio-physical (slope, soil quality) or socioeconomic (poverty, market access). Instead, theory could be a better (deductive) guide for identifying the constraints that bind and the contexts that matter, and for generating testable hypotheses. Theory could also help identify key structural parameters needed to forecast impacts in other contexts (Timmins and Schlenker, 2009).

Additionally, theory could help us think more generally about economies or diseconomies of scale. For example, a large-scale conservation policy can cause so-called general equilibrium effects, where interactions and feedback effects can generate complex dynamics. Such effects are more common when there is high dependence on natural resources in closely coupled human-natural systems (see later). For example, low crop prices may increase enrolment in PES schemes. However, if much farmland becomes enrolled in the conservation programme, the supply of crops may decrease, increasing the crop prices and affecting the number and types of parcels in the conservation programme.

Ross et al. (2010) use theory to build a dynamic computable general equilibrium (CGE) model with which they simulate the environmental and economic impacts of PES in Costa Rica. They find small general equilibrium effects, which gives us greater confidence in the empirical studies using observational data, which assume that no such effects exist.

Coupled systems

People and their environment are parts of dynamic coupled systems: the ecosystem structure and function impacts communities and people, whose use of natural resources in turn impacts ecosystem functioning (Dasgupta and Mäler, 2003). Perverse links persist and externalities abound because market and non-market signals (e.g. state and community institutions) often fail to emerge to correct the problem. By restricting or transforming natural resource extraction, conservation programmes will trigger a new dynamic in these coupled systems. Therefore, the nature of the coupling should influence how we model causal effects and what data we collect. First, it implies that we should consider joint economic and environmental outcomes. Second, it suggests that we should collect data on, and model the influence of, initial conditions (e.g. socio-political and bio-physical factors). However, few rigorous evaluations consider the joint outcomes of conservation programmes and model them as a non-linear function of initial conditions (Ferraro et al., 2011; Ferraro and Hanauer, 2011). Alternatively, analysts can evaluate programmes in coupled systems by conducting theory-based simulations. For example, Pattanayak et al. (2009) apply a dynamic CGE model to examine PA impacts in Brazil. They explicitly model how PAs reduce land available for agriculture and increase labour supply (because of lower levels of mosquito-borne diseases caused by deforestation). These land and labour market effects in turn impact deforestation.

In developing countries, the people–environment coupling is strong (Barrett et al., 2011). Environment-poverty trap theories suggest that small initial differences in the local context (e.g. either prior to or resulting from a conservation intervention) can cause large divergences in well-being and ecosystem functioning over time (Dasgupta and Mäler, 2003). Traps emerge partly because persistent poverty and rising disparities in each period make it difficult to generate (a) critical levels of investment for growth; and (b) conditions for good institutions to evolve and succeed (Dasgupta, 2009). Given these complex and multiple causes, the long-term impacts of a conservation programme can be very different from the short-term impacts.[11] Thus,

[11] From a cost-effectiveness perspective, programmes that ignore long-run impacts may be highly cost-ineffective. This is because many of the adverse long-run impacts could be irreversible or sticky (e.g. even if outcomes are somewhat reversible, the coupled system displays hysteresis).

where possible, we should collect data during and beyond the programme/project cycle. If long-run evaluations are impractical, Carvalho and White (2004) suggest using theory to describe a step-by-step sequence of causes and effects, collecting data on the initial steps and then examining how well each step is borne out during the project cycle. For example, researchers could check if social capital and local monitoring improve during the course of decentralization to signal the likelihood of long-run success.

12.4.2 Better methods

Impact evaluation is a rapidly evolving field. A parallel evolution can also be seen in the empirical conservation policy science literature. The earliest studies used simple before–after estimators or with–without estimators, without taking into account confounding factors that vary over time and space. The next generation relied on simple comparisons of the average difference in the outcomes between the matched treated and control observations. The most advanced, recent papers combine matching methods with bias-adjustment techniques and adjusted variance estimators (Abadie and Imbens, 2006, 2011; Imbens and Wooldridge, 2009) to examine average impacts as well as their heterogeneity (e.g. Ferraro et al., 2011). They also examine the robustness of the estimates to changes in the assumptions required for causal inference.

Sensitivity to identification assumptions

Because of the observational nature of the data used in quasi-experimental evaluations, concerns that an important confounding variable has been omitted, and thus the estimator used is biased, can never be eliminated. However, there are at least three approaches to considering how robust our inferences are to the presence of hidden bias: sensitivity analyses and partial identification. Sensitivity analyses start with the assumption that there is no hidden bias in the analysis and then progressively weakens this assumption and watches how the confidence interval of the estimate changes (Rosenbaum, 2002; DiPrete and Gangl, 2004; Altonji et al., 2005). For example, Andam et al. (2008) use sensitivity analyses to show that if an unobserved variable that strongly affects deforestation also increased the odds ratio of protection to differ between protected and unprotected plots by as much as 2.15, the 99 per cent confidence interval would still exclude zero (i.e. the results are robust to moderate hidden bias). Partial identification works in the reverse direction and starts with the weakest assumptions and gradually strengthens them while observing how the bounds on the causal impact change (Manski and Nagin, 1998). For example, Arriagada et al. (2012) show that if one were only to assume that (i) accepting payments to stop deforesting cannot induce a farmer to clear more forest; (ii) farmers who sign up for the programme have lower-than-average deforestation rates in the absence of the payment (i.e. positive self-selection); and (iii) a farmer is constrained to only clear all of the farm's forest or let the entire farm become forest, one could constrain the estimate of the impact of the PES programme to between 0 and 12 ha of additional forest per farm. A final way to eliminate rival explanations that stem from violations of key assumptions is to conduct tests of known effects (Rosenbaum, 2002). Using elaborate theory, one can identify the implications of violations in the underlying assumptions and test for evidence of these violations. For example, Sims (2010) hypothesizes that differential migration in regions with and without PAs could be causing changes in deforestation. Contrary to this hypothesis, she shows that migration patterns did not change during her study period.

Spillovers

The validity of the impact estimators also rests on the independence of the treatment and control observations. However, as previous studies have suggested, spillovers from 'treatment' to 'control' areas may violate the assumption. These spillovers may be negative (e.g. leakage; displacement of threat) or positive (e.g. increased enforcement or information flows). Although the literature has suggested excluding the potentially contaminated observations from the control group and explicitly testing for the presence of spillover effects at various distances, spillover analysis is not the norm. Additionally, as suggested in the discussion on feedbacks, these spillovers may be interesting phenomena that deserve direct modelling, instead of being treated as a nuisance to be dealt with.

Continuous, not discrete

Most impact evaluation studies have employed *discrete* treatments and covariates to examine how the impacts of protection vary across time, space, and intervention. However, programme data such as duration, area covered, and amount and the covariates that influence their impacts (e.g. slope, poverty rates, distance to markets) are all continuous variables.[12] Future evaluations could shift from answering whether the intervention has an impact on examining the overall shape of the production function—that is, the shape of the relationship is between the impact and the *continuous* treatment. Currently, only two studies have looked at the impacts of a conservation intervention as continuous functions of exposure: Sims (2010) considers the percentage of the

[12] The current practice employs some ad hoc rules and subjective decisions as to what constitutes a treated unit. For example, if only a part of a unit falls within a PA, then it is up to the researchers to decide whether to consider it protected.

locality that is a PA, whereas Arriagada et al. (2008) use a generalized propensity score method to examine how the density of PES contracts in a region affects deforestation. A similar suggestion applies to baseline covariates that modify the impact of a conservation programme: the modification may be continuous (as considered by Ferraro et al., 2011, and Nelson and Chomitz, 2011), and not discrete.

12.4.3 Better data

As noted by Ferraro and Pattanayak (2006), impact evaluations are constrained by inadequate data. This inadequacy exists because (a) most conservation interventions in poor countries are framed as independent proofs of concepts; (b) there is poor infrastructure, training, and history of systematic data collection for estimating causal effects; and (c) there are challenges of combining ecological, socioeconomic, and institutional data, all of which are needed for credible impact evaluations. Here we highlight two specific concerns.

Missing baselines

The availability of multi-period geospatial data with relatively fine resolution has allowed for deforestation patterns to be examined through time. Unfortunately, we have no such comparable repository of social-political data. We lack good baseline data on formal and informal institutions, the degree of information asymmetries, market access, intrinsic incentives and norms, and previous participation in forestry programmes (Jack et al., 2008; Pattanayak et al., 2010; Arriagada et al., 2012, Ferraro et al., 2012). Clearly, we need more and better socioeconomic and institutional data from biodiversity-relevant locations. Alternatively, we should be tailoring our sampling such as Baland et al. (2010) to address hard-to-obtain baseline characteristics.

Interdisciplinarity

Biodiversity is affected by both the amount and the structure of habitats (Krebs, 2001; Turner et al., 2001). Yet the data available are typically inadequate for measuring habitat quantity and quality. The outcomes researchers can study (e.g. deforestation, fires) are not necessarily the outcomes researchers wish to study (e.g. ecosystem structure and function).[13] Micro

[13] To highlight the importance of establishing interdisciplinary partnerships between economists and natural scientists, we focus on the ecological significance of the geospatial data rather than on the technical quality of the available datasets. The latter can be a significant hurdle to

studies of forest decentralization policies provide the few exceptions: fuelwood collected (Heltberg, 2001; Edmonds, 2002), percentage canopy cover per pixel (Somanathan et al., 2009), degree of lopping, presence of saplings, DBH, and canopy cover (Baland et al., 2010). While these metrics provide significant improvements over binary geospatial measures like 'forest–not forest', how these improved metrics relate to the ecosystem structure and function is unclear. For example, number of saplings in Baland et al. (2010) can be interpreted as a measure of the degree of forest regeneration, or may be used as an indicator of disturbance.[14]

To ensure that we are collecting the relevant data for informative impact evaluations, interdisciplinary partnerships between social, natural, and physical scientists are needed. The abundant data that natural and physical scientists collect are not amenable for use in rigorous impact evaluations. Partnerships between scientists with training in appropriate empirical designs for causal inference and scientists who understand what data are relevant and how to collect them are critical. Within a rigorous analytic framework, these partnerships can help identify the appropriate spatial and temporal scale(s) of the analysis in terms of both the socioeconomic and ecological processes, and select the appropriate proxies and metrics for biodiversity and ecosystem function, including measures of habitat connectivity and fragmentation.

12.5 CONCLUSION

Our review confirms previous claims that causal evidence on the effectiveness of conservation approaches commonly used in developing countries is rare (Ferraro and Pattanayak, 2006; Carpenter et al., 2009). The limited evidence suggests that PAs cause modest reductions in deforestation and thus may positively affect biodiversity. However, the evidence base for other environmental or social effects of PAs is much weaker, as is the evidence base for the environmental and social effects of PES, decentralization, and other popular conservation interventions. Because the geographic overlap between where

good impact evaluation studies as well: the presence of clouds, especially in tropical forests, the inability to distinguish between forest degradation and deforestation on one hand, and between different types of tree species on the other, as well as the lack of good metadata describing the methodology through which the geospatial datasets were obtained and the land-use categories classified, can significantly lower the reliability of geospatial datasets.

[14] The presence of saplings does not seem sufficient as these can be of invasive species that usually fare very well and grow very fast in disturbed areas; disturbance may actually result in changing the composition of the forest towards something that is no good for biodiversity conservation (Krebs, 2001).

PAs, PES, and decentralization are studied is limited, we cannot compare the relative effectiveness of these three approaches. In short, despite progress in the last six years in the empirical evaluations of conservation programme impacts, the evidence base—limited to a handful of tables—is simply too thin to say anything meaningful about the impacts of the billions of dollars invested in protecting biodiversity over the past five decades.

To deepen the conservation evidence base over the next ten years, we call for a programme of research—*Conservation Evaluation 2.0*—that focuses on four key goals: (1) clarifying the hypothesized causal pathways that connect conservation interventions through mechanisms to outcomes (so-called theories of change); (2) moving beyond estimates of average effects to an understanding of what factors moderate these effects (i.e. what leads to heterogeneous impacts?), what mechanisms are most important (i.e. estimates of mechanism causal effects), and what unintended consequences our programmes have (e.g. spatial spillovers); (3) diversifying the number of interventions and outcomes studied (e.g. studying social and environmental effects jointly; moving beyond just looking at effects on deforestation; unpacking concepts like 'protected' or 'decentralization'), and expanding the number of biodiversity-relevant locations studied; and (4) collecting data on costs, with which we can eventually compare the cost-effectiveness of different interventions across contexts and outcomes (and maybe even do cost–benefit analyses). In brief, *Conservation Evaluation 2.0* seeks better theory, better methods, and better data to swell the number and quality of rigorous impact evaluation studies in the conservation science literature.

Achieving the goals of *Conservation Evaluation 2.0* is hampered by the fact that few environmental policies and programmes are designed with impact evaluation in mind. We fail to implement our programmes or collect data in ways that facilitate credible impact estimates. Rather than hope we can find the relevant data and conditions to understand causal effects, heterogeneity, and mechanisms, we need to design policies and programmes with the explicit intent to measure their environmental and social effects. We thus urge practitioners and scholars to implement more programmes with experimental and quasi-experimental designs. Strong candidates for experimental designs include programmes targeted to individuals, firms, local communities, or municipalities. Particularly appropriate would be pilot programmes or programmes implemented by non-governmental organization partners, which are not subject to the conflicting agendas of the various stakeholders, and may have more flexibility with regard to where and with whom they operate. Whether or not experimental or quasi-experimental designs are used, good baseline data on the relevant socioeconomic and environmental factors are important for credible evaluations. Moreover, in order to ensure that such evaluations use the right data at the appropriate scale of analysis, and can credibly estimate both socioeconomic and environmental impacts, interdisciplinary partnerships between social and natural scientists are needed.

One reason why experimental and quasi-experimental designs are not the norm in conservation science is the perceived high costs of implementation (Ferraro and Pattanayak, 2006). We argue that the benefits of *Conservation Evaluation 2.0* exceed the costs because the information it provides will help (1) identify and discontinue programmes for which the desired causal impacts cannot be detected; (2) improve the cost-effectiveness of existing programmes; and (3) spur innovation. A cost–benefit analysis would also imply that evaluation is most fruitfully applied to commonly used policies, like the three we review, and to policies and programmes that provide an opportunity to test fundamental behavioural questions such as: how do land-users respond to financial incentives? How do local government decision-makers respond to information or capacity building? Evaluations focused on answering such fundamental behavioural questions are less about testing whether a specific project 'worked' and more about providing insights about the validity of the implicit and explicit causal models that underlie the global conservation investment portfolio.

We are not suggesting that all studies employ experimental and quasi-experimental designs. In fact, we believe many contexts will not be appropriate for them. Nevertheless, opportunities to use them exist, and without them, the evidence base will remain inadequate to guide our actions. No other evaluation designs offer as much power to identify the impacts of our policies and programmes by eliminating rival explanations for observed patterns of the environmental and social data we collect. But to apply these designs more broadly, conservationists, policy-makers, activists, and scholars need to invest in developing the requisite expertise.

In conclusion, our review highlights the paucity of causal evidence on the effectiveness of common conservation approaches. We urgently need more basic evaluations of average social and environmental impacts of common programmes from many more biodiversity-relevant locations (*Conservation Evaluation 1.0*). But we also need to move beyond these basic evaluations to a more advanced *Conservation Evaluation 2.0* research programme that seeks to measure how programme impacts vary by socio-political and bio-physical context, to track economic and environmental impacts jointly, to identify spatial spillover effects on untargeted areas, and to use theories of change to characterize causal mechanisms that can guide the collection of data and the interpretation of results. Only then can we usefully contribute to the debate over how to protect biodiversity in developing countries.

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13

Are Investments to Promote Biodiversity Conservation and Ecosystem Services Aligned?

*Stephen Polasky, Kris Johnson, Bonnie Keeler, Kent Kovacs, Erik Nelson, Derric Pennington, Andrew J. Plantinga and John Withey**

13.1 INTRODUCTION

Economists are used to thinking about maximizing an objective function subject to constraints. Individuals maximize utility subject to a budget constraint and perfectly competitive firms maximize profits given technology and prices. Though far less common, such thinking can also be applied to biodiversity conservation and environmental management. For example, several papers have analysed the objective of maximizing the number of species conserved through habitat protection given limited resources (e.g. Ando et al., 1998; Wilson et al., 2006; Murdoch et al., 2007). Applying an economic approach to conservation and environmental management requires stating a clear objective. In the conservation realm, however, there is not universal agreement on the objective:

> As a society, we have not even come close to defining what is the objective . . . We have to make up our minds here what it is we are optimizing. This is the essential problem confounding the preservation of biodiversity today. (Metrick and Weitzman, 1998, p. 21)

At present, there is a deep divide within the conservation community about the proper objective for conservation (Mace et al., 2012; Reyers et al., 2012).

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One school of thought focuses on ecosystem services and emphasizes the value of conserving biodiversity and ecosystems to provide ecosystem services that contribute to human well-being (Daily, 1997; MA, 2005; TEEB, 2010; Kareiva et al., 2011). Some prominent conservation organizations have adopted this approach. For example, the vision statement of the Convention on Biological Diversity (CBD) Strategic Plan for Biodiversity 2011–2020 is that conserving biodiversity and ecosystems is important for 'maintaining ecosystem services, sustaining a healthy planet and delivering benefits essential for all people' (CBD, 2010). The ecosystem services approach to conservation is consistent with a welfare economic approach that seeks to maximize social net benefits, where benefits include the contributions of ecosystems to human well-being. Of course, there are also benefits beyond ecosystem services, so that maximizing well-being and maximizing the value of ecosystem services are not synonymous. This approach requires integrated ecological-economic modelling that demonstrates the link between ecosystem management, ecological processes, the provision of ecosystem services, and consequent impacts on human well-being (Daily et al., 2009; NRC, 2005).

A second school of thought is that conservation should be based on ethical arguments about the intrinsic value of nature (Rolston, 1988; McCauley, 2006; Redford and Adams, 2009; Vira and Adams, 2009). In this school of thought, biodiversity is to be conserved for its own sake, regardless of whether or not it contributes to human well-being (Ehrenfeld, 1988). This does not mean that biodiversity does not also contribute to human well-being, but that the motivation for conservation comes from the intrinsic value of nature. The intrinsic value of nature motivation for conservation represents a fundamental departure from a welfare economics perspective, where nature has instrumental value (i.e. it contributes to human well-being). Under the intrinsic value of nature approach, biodiversity conservation is an ethical obligation and should occur even when doing so imposes burdens upon society that reduce human well-being. We revisit the relationship between ecosystem services, biodiversity, and human well-being in the discussion section.

Though the divisions between focusing on human well-being versus focusing on the intrinsic value of nature are deep in terms of underlying philosophy, do they matter in a practical sense in terms of land use or resource allocation? Do management decisions of a conservation agency aimed at maximizing the value of ecosystem services differ dramatically from management decisions aimed at conserving biodiversity? If management prescriptions from these two approaches closely align, then we would argue that conservation planners can proceed without worrying too much about the underlying philosophical debates. If this is the case, disputes over the proper goal for conservation would be yet another example of the famous saying about debates in academia: 'The politics of the university are so intense because the stakes are so low', which is also known as Sayre's Law (Shapiro, 2006, p. 670). If, on the other hand, management prescriptions do not closely align, then conservation managers will need to address the question of the proper objective function before deciding what actions to take.

In this chapter, we address the degree of alignment between ecosystem services and biodiversity conservation using data from the state of Minnesota, US. In 2008, Minnesota voters passed the Clean Water, Land and Legacy Amendment (Legacy Amendment). The Legacy Amendment increased the state sales tax by three-eighths of 1 per cent for twenty-five years, likely raising more than $250m per year. Of these funds, 33 per cent are allocated to a Clean Water Fund to conserve and enhance water quality, and 33 per cent are dedicated to an Outdoor Heritage Fund to protect and restore prairies, forests, wetlands, and other wildlife habitat. Together these two funds will provide an estimated $171 million annually for conservation in Minnesota. We analyse whether using the Legacy Amendment Funds towards a strategy that aims to maximize the value of ecosystem services will choose similar land for conservation compared with a strategy that aims to maximize the conservation of biodiversity.

While it would be ideal to include the value of all ecosystem services and all biodiversity, doing so is well beyond current capabilities. Here we model the provision and value of carbon sequestration and the reduction of phosphorus in water bodies, the latter being the most important factor for surface water quality in the state, which closely matches with the goals of the Clean Water Fund. As our measure of biodiversity we use the predicted occurrences of vertebrates (including breeding, game, and listed species) because their distributions and associations with land use and land cover are well known compared with other organisms, and match the general goals for the Outdoor Heritage Fund.

We compare the ecosystem service and biodiversity strategies on a static landscape in which the only change in land use is brought about by purchase of land for conservation ('static analysis'), and on a more realistic case in which conservation occurs amidst the backdrop of other land-use change ('dynamic analysis'). We find that in both static and dynamic analyses purchasing land for one objective has a positive effect on the other objective, but that the alignment of objectives is far from perfect. In the case of static land use, targeting ecosystem services generated a biodiversity score that was 53 per cent of the maximum score obtained when targeting biodiversity. When we targeted biodiversity we generated a value of ecosystem services that was 70 per cent of the maximum value of ecosystem services obtained when targeting services. In the dynamic land-use case, targeting ecosystem services generated 47 per cent of the biodiversity score as compared with targeting biodiversity, and targeting biodiversity generated 65 per cent of the value of ecosystem services as compared with targeting services.

Most prior work looking at the spatial pattern of the provision of bundles of ecosystem services and biodiversity describes the degree of spatial correlation given the current pattern of land use (e.g. Chan et al., 2006; Egoh et al., 2008, 2009; Naidoo et al., 2008; Raudsepp-Hearne et al., 2010). Different land uses generate different bundles of services. For example, intensive agricultural production is associated with high production of agricultural products but low water quality and carbon storage, while conserved forested areas often have high carbon storage, habitat, and recreation value but low commercial returns. In this chapter, we address the more policy relevant question of how to maximize the increase in the provision of ecosystem services or biodiversity conservation through changes in land use for a given cost measured as the value of the land and the restoration to potential vegetation land cover. The closest prior papers in this vein are Naidoo and Ricketts (2006), Nelson et al. (2008, 2009), and Polasky et al. (2008, 2011). Apart from Egoh et al. (2010) these papers do not directly address the question of alignment between ecosystem service objectives and biodiversity conservation objectives.

We describe the data and models used to perform this analysis in section 13.2. Results are presented in section 13.3. Section 13.4 contains a brief summary of major findings as well as comparisons of our results to prior work in a similar vein. We conclude section 13.4 with a discussion of outstanding issues that require further research.

13.2 DATA AND METHODS

We model two important drivers of changes in ecosystem services, carbon sequestration and water quality, and the provision of habitat for biodiversity under alternative land-use scenarios and decision-making criteria. We compare the outcome of land acquisition for conservation guided by an ecosystem service objective with land acquisition guided by a biodiversity objective. We compare these objectives under an assumption of static land use and under a dynamic land-use change model. We begin this section by describing the land-use and land-cover data. Next we describe the land-use scenarios evaluated in this chapter. We then discuss the models used to quantify carbon storage, water quality, and habitat for biodiversity. We discuss the opportunity cost and restoration cost of conserving land. Finally, we explain the optimization framework to guide conservation strategies that incorporates costs, biodiversity, and ecosystem service benefits.

13.2.1 Land-use and land-cover data

We used a baseline 2001 land-use map and predicted 2026 land-use maps to examine how conservation funds should be allocated during the 25-year period over which conservation funds are available. We generated the baseline 30-metre resolution land-use map by downloading the 2001 National Land Cover Database (NLCD) for Minnesota (Homer et al., 2007). We converted all NLCD land covers into one of five general land-use types (cropland, pasture, range, forest, and urban) using conversions shown in Table A1 of the Appendix.[1] We used data from a national map of private and public lands (Conservation Biology Institute, 2010) to delineate public and private lands within the state.

Ecosystem services, habitat quality, and land values were calculated at the sub-county unit level, using boundaries defined by the Minnesota Department of Revenue for purposes of property tax reporting. For most of the state, sub-county units are townships, except in the north where townships are quite large and sub-county units are defined on a smaller area more closely resembling the size of townships in the rest of the state. This spatial delineation allows use of the greatest detail on land costs across the state, especially in the largely undeveloped northern region.

Land close to streams and rivers generally has more direct impact on water quality (Osborne and Kovacic, 1993). To distinguish between the ecosystem service benefits of conserving lands close to surface water bodies, we used 100-metre buffers around centrelines for fifty-two major rivers in the state (Minnesota Department of Natural Resources, 2012).

Combining information from these various data layers, we created maps identifying the area of each land-use type (cropland, forest, pasture, urban, and range) both within and outside of the 100-metre buffers, held in both public and private ownership, for every sub-county unit in Minnesota as of 2001.

13.2.2 Land-use scenarios

We analysed optimal land acquisition for conservation under static and dynamic land-use scenarios. In both scenarios, land acquired and conserved was assigned its sub-county potential vegetation proportional mix of native land covers: forest, prairie, or wetland. The native land cover proportion was based on the LANDFIRE biophysical settings layer (NatureServe, 2009), which assigns an ecological system code to each 30-metre pixel based on potential vegetation and natural disturbance regimes. In both static and dynamic land-use scenarios, we also assumed that management of public lands remained unchanged and that all conservation funds were used to acquire and conserve lands that were private in 2001.

[1] A detailed Appendix to this chapter is available online at <http://oxrep.oxfordjournals.org/cgi/contentembargo/full/grs011/DC1>.

In the static land-use scenario, private land that was not chosen for conservation remained in its 2001 land use. In the dynamic land-use scenario, we projected land use to 2026 for private lands that were not conserved using a land-use change matrix. The land-use change matrix gives the probabilities of transitions from one land use to another over the 25-year period between 2001 and 2026 for each sub-county unit and for lands within and outside the 100-metre water buffers. Because the land use in 2026 is probabilistic, we chose a particular land use for each hectare of private land using a random number generator and the appropriate 25-year land-use change matrix for the hectare. By simulating the choice for each hectare, we determined the distribution of land use in 2026 by land-use type within each sub-county unit within and outside the 100-metre water buffers. This procedure was repeated 100 times for each conservation strategy, generating 100 maps of 2026 land use on private land, which we then combined with conservation and public land to generate 100 state-wide land-use maps for 2026 under the conservation strategy.

These land-use transition matrices are described in more detail in Radeloff et al. (2012). In that research, five-year land-use change probabilities were used to simulate land-use change across the US. We used the land-use change probability matrices specific to Minnesota for five sequential five-year periods to get the cumulative sum of changes over twenty-five years in the state. Unlike Radeloff et al. (2012), we have modified the transition matrices to take into account the effect of land market price feedbacks on the transition probabilities, as in Lubowski et al. (2006).

13.2.3 Carbon storage and sequestration

Although climate change mitigation was not an explicit goal of the Legacy Amendment, state policies establish aggressive greenhouse gas emission reduction targets and identify biologic sequestration of carbon as an important strategy to achieve these goals (MCCAG, 2008). We calculated carbon storage values for soil and for biomass for each land-use type in each sub-county unit in Minnesota. We estimated carbon sequestration that would be achieved under a conservation strategy by calculating the differences in carbon storage under the strategy relative to the 2001 baseline.

To calculate the quantity of carbon stored in soils we used a national map of soil carbon (Sundquist et al., 2009) combined with land cover and county boundary data to generate average soil carbon storage values for each land-use type in each county in Minnesota. We did not have data on soil carbon in wetlands. Wetlands generally have some of the higher soil carbon levels. Therefore, we used the highest observed soil carbon level for land-use types for wetlands.

We also calculated carbon storage in above-ground biomass. Because biomass from cropland, pasture, and range is generally harvested and removed each year, we assumed that each of these land-use covers stored zero above-ground carbon in biomass. To calculate biomass carbon on forest hectares we used Forest Inventory Analysis (FIA) data and Smith et al. (2006). The FIA dataset indicates the proportion of forest land in each county in the forest types oak-hickory, white-red-jack pine, and aspen-birch. We distinguished between management of private forest land and conservation forest land. For forests on land that was set aside under a conservation strategy we assumed that restored forest would attain biomass carbon levels of a 95-year-old forest with the county's mix of forest types. Smith et al. (2006) provide carbon storage values for each major forest type by forest age class. We assumed that private forest land was in managed rotations, where trees were harvested at specified age (the Faustmann rotation age), and steady-state harvesting maintains a constant proportion of land in each age class up to the rotation age. For Minnesota forests the Faustmann rotation age was between thirty and sixty years. Again, Smith et al. (2006) was used to find biomass carbon levels associated with tree ages and a county's mix of forest types to determine a private forest hectare's biomass carbon levels. Finally, we assume that an urban hectare in a county has one-tenth of the above-ground biomass carbon of a private forest hectare in the same county.

We calculated monetary values of the changes in carbon storage using estimates of the social cost of carbon (Tol, 2009). The social cost of carbon is the cost to society incurred by the potential climate change damages from each additional tonne of carbon emitted to the atmosphere. Values for the social cost of carbon reported in the literature range from near \$0 to over \$500 per ton of carbon (Tol, 2009). In this chapter, we used a base-case estimate of \$126.40 per ton carbon (\$34.47 per ton of carbon dioxide (CO_2)) in constant 2011 dollars, based on a value of \$91 in 1995 constant dollars for the median fitted distribution for social cost of assuming a 1 per cent pure rate of time preference (Tol, 2009).

13.2.4 Water quality: phosphorus retention

A core goal of the Legacy Amendment is to protect and restore water quality in Minnesota. Land use can impact water quality by contributing sediment, nutrients, or other pollution to surface and ground water. Conserved lands can provide an important ecosystem service by capturing polluting nutrients and sediment before they reach adjacent water bodies. In this analysis, we focus on phosphorus pollution, which is the leading cause of surface water impairment in the upper Midwest (Carpenter et al., 1998). We used the InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs; Tallis et al., 2010, <http://invest.ecoinformatics.org/>) water models to estimate the water-quality benefits provided by land acquisition and conservation. InVEST is a spatially explicit model that applies a two-step process to determine the influence of land cover on water quality. First, the model calculates the average annual water yield in each mapped grid cell using climate data, geomorphological information, and land-use and land-cover (LULC) characteristics. The model does not incorporate sub-surface or ground water flows but assumes that all precipitation not lost to evapotranspiration goes to surface water runoff. In the second step, water yield is combined with information about phosphorus loading and the phosphorus retention capacities of each LULC type to calculate the annual phosphorus exports from each grid cell. Phosphorus exports from cells are routed via surface water flows to other cells, where some of the phosphorus may be filtered or additional phosphorus added, until the surface water flows into a water body. Once phosphorus reaches a water body the model assumes no additional retention, or removal occurs before delivery to the mouth of the watershed.

We used the InVEST water models to calculate the phosphorus loading for the 2001 baseline map. Because the InVEST water models are spatial and rely on surface water flows to route nutrients, we ran the models using the eighty-one eight-digit hydrologic unit code (HUC) basins in Minnesota. Average phosphorus loadings were then assigned to each sub-county unit based upon the location of the sub-county unit within the HUC basins. We also used the 2001 baseline map to calibrate the average per-hectare phosphorus loading and phosphorus retention capacities of each LULC type, both outside and inside of the 100-metre buffers around rivers and streams. Reductions in phosphorus loading to be achieved in 2026 by conservation decisions were calculated by multiplying the LULC proportions adjusted following land acquisition by the average per-hectare loadings for each LULC type. The percentage change in the loadings of each basin was calculated by finding the difference of the loadings associated with the LULC change divided by the total baseline loading of phosphorus to the basin. Basins further downstream of where LULC change occurs also experience a change in water quality.

We used a national meta-analysis conducted by Johnston et al. (2005) to generate estimated annual per-household willingness-to-pay (WTP) values for improved water quality. Following the guidelines in Johnston and Besedin (2009) we adapted parameters in the WTP function to reflect appropriate geographic area, water body type, and mean household income. The model estimates WTP as a function of changes in water quality relative to baseline conditions, with water quality described by the Resources for the Future (RFF) water quality ladder. The RFF water quality ladder links changes in water uses (drinking, boating, swimming, and fishing) to variations in biophysical characteristics (dissolved oxygen, turbidity, pH), and uses a qualitative point system to represent changes in the value of uses that correspond to changing water quality (Carson and Mitchell, 1993). To establish baseline water quality for each HUC basin, we obtained statewide data on lake trophic state index (TSI; Carlson, 1977) from the Minnesota Pollution Control Agency (MPCA; personal communication with Steven Heiskary, 2012). We then mapped average TSI values for lakes within each HUC basin to the RFF water quality ladder. Based on consultation with local water-quality experts, we assumed that a 50 per cent reduction in phosphorus loading relates to a two-point increase along the RFF water quality ladder. Combining these water-quality parameters with the Johnston et al. (2005) WTP function, we generated estimates of annual WTP for the 50 per cent reduction of from \$24.97 to \$44.72 per household in 2011 constant dollars. The values were prorated to the per cent change in phosphorus loadings modelled by InVEST; for example, for a WTP value of \$10 per household for a 50 per cent reduction, a 1 per cent reduction in phosphorus loadings was prorated to \$0.20. The prorated WTP per household is an annual value which we then converted into a present value of benefits assuming permanent water-quality improvement. The future benefits of public goods should be discounted at a rate close to the market rate of return for risk-free financial assets (Howarth, 2009), which we assumed to be 2 per cent. The present value of WTP values per household for each basin is multiplied by the number of households per sub-county unit, based on the average of the number of households in 2010 and population projections for 2025 (Minnesota Department of Administration, 2007).

13.2.5 Habitat for biodiversity

The Legacy Amendment also directs funds to protect ‘fish, game and wildlife habitat’ and has an explicit goal of enhancing Minnesota’s capacity to conserve and enhance biological diversity. We model the baseline 2001 map and the 2026 alternative land-use scenarios to compare the potential benefits for biodiversity of alternative conservation strategies. The biodiversity model evaluates the potential for different land cover types in a sub-county unit to provide habitat for a set of vertebrate species based on current distributions and habitat associations. First, we estimate total vertebrate species richness for each LULC type at the sub-county unit level. We use information on the current predicted distribution of individual species based on actual habitat characteristics within their general ranges, as determined by the Minnesota Gap Analysis Project (MN-GAP; Drotts et al., 2007). MN-GAP includes species found in Minnesota that are listed as: breeding, state endangered or threatened, of special conservation concern, a fur-bearer, big game, small game, or migratory game bird. MN-GAP includes 354 vertebrate species (21 amphibians, 28 reptiles, 75 mammals, and 230 birds). For sub-county units that did not contain a given LULC type, we determined county-level richness estimates and used these to substitute for the missing sub-county unit-level LULC type. We determine the habitat for biodiversity score for a sub-county unit by multiplying the species per LULC type estimate by its corresponding LULC area for the total of public, private, and conserved lands, and summed this score across all LULC types. This sum produces the number of habitat units in the sub-county unit, which indicates the conservation value of those lands to support these species. Higher scores indicate more available habitat to support more species, and therefore sub-county units with greater value for biodiversity conservation. We then sum the sub-county unit scores across all units to generate a score for the entire state under a given conservation strategy.

13.2.6 Conservation budget and opportunity costs

We created a conservation budget by combining the two largest allocations of the Legacy Amendment, the Clean Water Fund dedicated to improving water quality, and the Outdoor Heritage Fund targeted to preservation of wildlife habitat, which generated $171 million per year. Assuming a 2 per cent real interest rate, the total present value of the conservation budget over the 25-year duration of the Amendment was $3.319 billion.

We downloaded recent land value data for private crop, timber, and pasture land uses in each sub-county unit in Minnesota (<www.landeconomics.umn.edu>). The statewide average land values for cropland, timberland, and pasture land are $24,989, $10,225, and $8,289 per hectare, respectively. Land values, however, vary by sub-county unit. We also used land-restoration costs to estimate the transition cost of shifting from one form of private land use to a conserved native land-use type (LSOHC, 2009). The restoration cost used for conserved wetland, forest, and prairie is $2,904, $3,743, and $2,629 per hectare, respectively. We were able to attain only state-wide average numbers for restoration costs so these did not vary by sub-county unit. We combined land value data that represent the opportunity cost of conserving land and land restoration costs to estimate the total costs of switching from private to conserved land.

13.2.7 Optimization for targeting conservation investment

Land-use conversion that changes land cover causes a change in the provision of ecosystem services and habitat for biodiversity. We used the carbon, water quality, and habitat for biodiversity models described earlier to define the change in the value of ecosystem services and biodiversity caused by a land-use change. For the static land-use scenario, the expected benefits of conservation are given by the gain in conservation score across the state of

Minnesota (biodiversity score or value of ecosystem services) generated with land acquisition and restoration to the potential vegetation natural state given baseline land uses in 2001. In the dynamic land-use scenario, acquisition for conservation also prevents land from converting to some other land use by 2026. The expected benefits of conservation for this scenario are given by the gain in conservation score across the state of Minnesota generated with land acquisition and restoration to the potential vegetation natural state *plus* the conservation score created by expected land-use change between 2021 and 2026 on land that remains private. The costs of conserving are the sum of the land cost plus the costs of restoring to the potential vegetation natural land cover.

We solved the static and dynamic land-use scenario problems for the ecosystem services and biodiversity objectives with the Generalized Algebraic Modeling System (GAMS) 23.5.1 using the linear programming solver CPLEX. The optimization routine finds the land conservation pattern that maximizes the expected increase in the value of ecosystem services or habitat for biodiversity given the budget constraint fixed by the amount of the Legacy Amendment Funds. The optimization model selects the land with the highest (expected) gain in conservation target value per dollar expended until the conservation budget is exhausted.

In the dynamic land-use scenario, we solved two optimization problems. In one solution, we assumed that the conservation planner was unaware of land-use change dynamics and planned as if the land use would remain constant at 2001 land use except for conservation acquisitions ('dynamic conservation solution ignoring land-use change in planning'). In the other solution, we assumed the conservation planner takes account of land-use change dynamics in choosing which lands to conserve ('dynamic conservation solution incorporating land-use change in planning'). In this case, it may be worthwhile conserving land not because it will increase ecosystem service or biodiversity values, but simply to prevent land-use conversion that may result in significant declines in values. The inclusion of threat of land-use conversion can mean the dynamic and static solutions can diverge significantly. The static model will select the land for conservation that generates the largest increase in returns per dollar compared with the 2001 baseline land use, while the dynamic model will select lands that generate the largest increase in returns per dollar compared with the projected distribution of land uses in 2026.

13.3 RESULTS

The main issue we address in this chapter is the degree of alignment between ecosystem services and biodiversity conservation strategies. We allocated the total conservation budget towards setting aside land, using a strategy to

Table 13.1. Change in the value of ecosystem services and the biodiversity score with a static landscape under an ecosystem service objective and a biodiversity objective

Objective	Ecosystem services ($m)	Biodiversity score (m)
Ecosystem service objective	9,026	5.58
Biodiversity objective	6,333	10.6

Notes: The value of ecosystem services is reported in millions of 2011 constant dollars. Biodiversity scores are reported in millions of habitat units (representing predicted richness × habitat area).

maximize the value of ecosystem services (carbon sequestration and phosphorus retention) and a strategy to maximize the value of habitat for biodiversity conservation (Table 13.1). Each strategy resulted in increases in both objectives. The optimal solution when targeting ecosystem services generated 53 per cent of the biodiversity score as compared with targeting biodiversity (5.58m units versus 10.60m—note that these are not units of area or species but a combination of both to indicate conservation value of lands). The optimal solution when targeting biodiversity generated 70 per cent of the value of ecosystem services as compared with targeting services ($6.333 billion versus $9.026 billion).

Under either strategy, the benefits of conservation outweigh the costs. We take the total costs of the conservation programme to be equal to the total conservation budget available, $3.319 billion, which is equal to the sum of expenditures on land purchase and restoration costs. Land purchase costs represent the opportunity cost of forgone returns when the land is put in conservation versus some other use that generates returns for the landowner. The increase in the value of ecosystem services is $9.026 billion for the case where we maximize ecosystem services, which yields a return on investment of $2.71 per dollar invested. In reality, there are likely to be transactions costs (expenses related to purchase, management, and administration) that would inflate the costs beyond the value of the land and restoration costs. If we assume that transactions costs add an additional 20 per cent to programme costs, then full programme costs would be $3.983 billion. In this case, the return on investment in conservation is $2.27 per dollar invested. Benefits far exceed costs, even though we include only the value of carbon sequestration and water-quality improvement but not the value of other ecosystem services or habitat conservation.

Land-use change from other factors will likely have more of an impact on future land use in terms of total land area than will setting aside land through purchase from conservation funding. Using the methods described in section 13.2, we predicted how land use would likely change over the period 2001–26. If there were no conservation strategy over this time period, the value of ecosystem services would rise by $8.245 billion, while the biodiversity score would fall by 6.7m (Table 13.2). These results are driven by the fact that

Table 13.2. Change in the value of ecosystem services (ES) and the biodiversity score between 2001 and 2026 for the State of Minnesota with a dynamic landscape

Scenario		Private lands					Conserved land					Total (private + conserved land)	
		Biomass carbon ($m)	Soil carbon ($m)	Water ($m)	ES ($m)	Biodiversity score (m)	Biomass carbon ($m)	Soil carbon ($m)	Water ($m)	ES ($m)	Biodiversity score (m)	ES ($m)	Biodiversity score (m)
No funds for conservation	Mean	3,802	3,919	525	8,245	−6.70	0	0	0	0	0	8,245	−6.70
	SD	5.50	4.62	2.19	9.54	0.01						9.54	0.01
Dynamic conservation solution incorporating land-use change in planning													
ES objective	Mean	3,273	3,880	491	7,644	−8.36	7,737	711	525	8,973	5.2	16,616	−3.17
	SD	4.20	2.08	2.75	7.65	0.01						7.65	0.01
Biodiversity objective	Mean	3,255	4,003	515	7,774	−9.31	5,166	493	216	5,875	10.1	13,650	0.82
	SD	5.17	4.24	2.10	8.30	0.01						8.30	0.01
Dynamic conservation solution ignoring land-use change in planning													
ES objective	Mean	3,211	3,846	485	7,542	−8.65	7,661	832	533	9,026	5.6	16,568	−3.07
	SD	4.35	4.19	2.18	7.69	0.01						7.69	0.01
Biodiversity objective	Mean	3,117	4,003	510	7,630	−10.14	5,521	553	259	6,333	10.6	13,963	0.47
	SD	5.08	4.87	1.91	9.04	0.01						8.30	0.01

Notes: We report changes with no funds for conservation, and with a conservation budget of $10.559 billion. With a conservation budget we report results for the case where the planner factors in potential land-use change into the optimization routine and where the planner ignores potential land-use change. All dollar figures are reported in millions of 2011 constant dollars. Biodiversity scores are reported in millions of habitat units (representing predicted richness × habitat area).

croplands are expected to decline by approximately 1.37m hectares. Forests are expected to have the largest net gain (0.63m hectares) followed by urban (0.37m hectares), range (0.26m hectares), and pasture (0.11m hectares). We show the conversion from each land use to each other land use under the no conservation strategy as well as the dynamic and static conservation strategies in Table 13.3. While there is considerable variation in both the value of ecosystem services and the biodiversity score within a given land-use type, on average cropland scores low in terms of both carbon sequestration and water quality relative to other land uses (Table 13.4). The movement out of croplands then tends to increase the value of ecosystem services generated. In terms of habitat value, however, croplands score relatively well, since many species use croplands for feeding or nesting (e.g. migratory water birds or open-land birds and mammals), especially those adjacent to water and wetlands. The movement out of croplands and into other types of land use results in a drop in the biodiversity score.

We then analysed how well aligned the ecosystem services and biodiversity strategies were against a backdrop of ongoing land-use change (Table 13.2). We analysed two planning strategies: (a) a dynamic conservation solution incorporating land-use change in planning, and (b) a dynamic conservation solution ignoring land-use change in planning. For both strategies, we again find that targeting ecosystem services also increases biodiversity conservation, and vice versa. For the dynamic conservation strategy incorporating land-use change in planning, the optimal solution when targeting ecosystem services generated 47 per cent of the biodiversity score as compared to targeting biodiversity. The gain in biodiversity under the ecosystem service strategy was from −6.70m to −3.17m units for an increase of 3.53m, whereas the biodiversity score increased to 0.82m under the biodiversity strategy, for an increase of 7.52m. The optimal solution when targeting biodiversity generated 65 per cent of the value of ecosystem services as compared with targeting services. The value of ecosystem services increased from $8.245 billion without the conservation programme to $13.650 billion with the biodiversity strategy for an increase of $5.405 billion, and $16.616 billion with the ecosystem services strategy for an increase of $8.371 billion. Taking account of land-use change over this time reduced the alignment of objectives by a small amount, but the general conclusion about the large degree of overlap remains.

The dynamic conservation strategy incorporating land-use change in planning should be superior to the strategy that ignores potential land-use change in planning. But how much improvement does this more sophisticated strategy yield? We found that incorporating land-use change in planning did only slightly better as compared to the strategy that ignored potential land-use changes in planning for both the ecosystem services objective ($16.616 billion versus $16.568 billion or 0.5 per cent higher) and the biodiversity conservation objective (7.52m versus 7.17m, or 4.9 per cent higher). Inclusion of land-use

Table 13.3. Average land-use change dynamics between 2001 and 2026 for Minnesota by land-use type

2001–2026	No funds for preservation	Dynamic conservation solution incorporating land-use change in planning		Dynamic conservation solution ignoring land-use change in planning	
		ES objective	Biodiversity objective	ES objective	Biodiversity objective
Cropland to cropland	62.3	61.9	62.1	61.9	62.2
Cropland to pasture	9.1	9.1	9.1	9.0	9.1
Cropland to forest	3.9	3.9	3.9	3.9	3.9
Cropland to urban	1.8	1.8	1.8	1.8	1.8
Cropland to range	3.1	3.0	3.0	3.0	3.1
Pasture to cropland	3.5	2.8	2.8	2.7	2.6
Pasture to pasture	7.1	5.7	5.4	5.6	5.0
Pasture to forest	4.7	3.7	3.5	3.5	3.2
Pasture to urban	0.8	0.7	0.6	0.6	0.6
Pasture to range	0.5	0.4	0.4	0.4	0.4
Forest to cropland	0.4	0.4	0.4	0.4	0.4
Forest to pasture	0.8	0.8	0.8	0.8	0.8
Forest to forest	26.5	26.1	25.9	26.2	26.1
Forest to urban	1.0	1.0	1.0	1.0	1.0
Forest to range	0.3	0.3	0.3	0.3	0.3
Urban to urban	9.5	9.5	9.5	9.5	9.5
Range to cropland	0.3	0.2	0.2	0.2	0.3
Range to pasture	0.7	0.5	0.6	0.6	0.6
Range to forest	0.2	0.2	0.2	0.2	0.2
Range to urban	0.1	0.1	0.1	0.1	0.1
Range to range	5.5	4.4	4.7	4.6	5.2

Notes: We report the amount of land in cropland, pasture, forest, urban, and rangeland that stayed in its initial use or converted to another land-use type between 2001 and 2026 for various conservation scenarios. All values are reported in hundred thousand hectares.

Table 13.4. Average impacts of land-use change on the value of ecosystem services and the biodiversity score by land-use category

2001 to 2026 land use	Water quality outside buffer ($/ha)	Water quality inside buffer ($/ha)	Soil carbon ($/ha)	Biomass carbon ($/ha)	Biodiversity (score/ha)
Cropland to cropland	0	0	0	0	0.00
Cropland to pasture	552	1,986	2,584	0	−8.16
Cropland to forest	969	2,950	2,239	5,430	−3.21
Cropland to urban	−148	−559	1,404	543	−9.06
Cropland to range	552	1,986	3,089	0	−3.27
Pasture to cropland	−552	−1,986	−2,584	0	8.16
Pasture to pasture	0	0	0	0	0.00
Pasture to forest	605	2,201	−345	5,430	4.95
Pasture to urban	−638	−2,226	−1,180	543	−0.90
Pasture to range	0	0	506	0	4.89
Forest to cropland	−969	−2,950	−2,239	−5,430	3.21
Forest to pasture	−605	−2,201	345	−5,430	−4.95
Forest to forest	0	0	0	0	0.00
Forest to urban	−1,036	−3,059	−835	−4,887	−5.85
Forest to range	−605	−2,201	850	−5,430	−0.06
Urban to urban	0	0	0	0	0.00
Range to cropland	−552	−1,986	−3,089	0	3.27
Range to pasture	0	0	−506	0	−4.89
Range to forest	605	2,201	−850	5,430	0.06
Range to urban	−651	−2,237	−1,685	543	−5.79
Range to range	0	0	0	0	0.00

change affects the overall outcome of biodiversity and ecosystem services but somewhat surprisingly taking this into account in planning had relatively little effect on strategy or expected outcomes.

The spatial pattern of the lands purchased for conservation under both ecosystem service and biodiversity conservation strategies for dynamic and static strategies is shown in Figure 13.1. The total amounts of land conserved by land-use category and the change in the value of ecosystem services and biodiversity score by land-use category with conservation are reported in Tables 13.5 and 13.6. In general, these patterns reflect the spatial distribution of the major land uses and vegetation biomes of Minnesota, with coniferous forest in the north-east, a mix of deciduous forest, croplands, and pasture in the central and south-east portions, and croplands and grassland in the west-central and south-west portions. Under the ecosystem services strategy, purchased lands were clustered in the forested north-east and the south-east, with very little land purchased in the western or central portions of the state. Lands converted from croplands, pasture, or range to conserved forest resulted in large increases in carbon sequestration, which dominated the value of ecosystem services under our baseline assumptions. The value of the increase in carbon storage made up \$15.6 billion of the \$16.616 billion increase in value of ecosystem services, with water-quality improvements making up just over \$1 billion. Under the biodiversity strategy purchased lands were spread throughout the entire state. Croplands and grasslands both had relatively high value for biodiversity, so conserving these land-cover types along with other lands added to the biodiversity score. The spatial pattern seen in the biodiversity strategy reflects the fact that we considered biodiversity in general terms, including both common and rare species, habitat generalists and specialists, and both residents and migratory (waterbird) species. Doing so gave value to protecting a broad range of habitats and locations. For example, conservation of grassland species requires conservation efforts in the west and south-western portions of the state, whereas forest species require conservation efforts in the north-east and south-east. The difference in spatial pattern of the land purchased for conservation under the different scenarios and different objectives is shown in Figure 13.2. In general, the dynamic land-use scenario puts a higher value on conserving land that may convert to a land use with lower conservation value, such as urban land use, and so conserves more land in regions with higher development pressure (Figures 13.2A and 13.2B). The ecosystem services objective with a heavy weight on carbon puts great value on restoring forest lands in the north-east and south-east and less weight on conservation in western parts of the state (Figure 13.2C).

There is considerable uncertainty about many of the biophysical relationships in the biodiversity and ecosystem services models, but probably even greater uncertainty exists about the proper values for carbon sequestration and water-quality improvement. We performed a sensitivity analysis on our results

Table 13.5. Hectares conserved between 2001 and 2026 by land-use category

2001–2026 land use	No funds for preservation	Dynamic conservation solution incorporating land-use change in planning		Dynamic conservation solution ignoring land-use change in planning	
		ES objective	Biodiversity objective	ES objective	Biodiversity objective
Cropland to conserved	0.00	0.45	0.26	0.48	0.13
Pasture to conserved	0.00	3.27	3.85	3.68	4.81
Forest to conserved	0.00	0.46	0.66	0.33	0.40
Range to conserved	0.00	1.31	0.99	1.02	0.35

Note: All values are reported in hundred thousand hectares.

Table 13.6. Average impacts of conserving land on the value of ecosystem services and the biodiversity score by land-use category

2001–2026 land use	Water service outside buffer ($/ha)	Water service inside buffer ($/ha)	Soil carbon ($/ha)	Biomass carbon ($/ha)	Biodiversity (score/ha)
Cropland to conserved forest	1,307	4,252	2,239	19,312	–2.32
Cropland to conserved grassland	392	1,275	2,584	0	–2.80
Cropland to conserved wetland	1,307	4,252	4,248	0	1.13
Cropland to conserved (average)	784	2,551	2,837	7,013	–1.11
Pasture to conserved forest	588	2,190	–345	19,312	5.84
Pasture to conserved grassland	177	657	0	0	5.36
Pasture to conserved wetland	588	2,190	1,664	0	9.29
Pasture to conserved (average)	353	1,314	253	7,013	7.05
Forest to conserved	1	2	598	1,819	2.10
Range to conserved forest	588	2,190	–850	19,312	0.95
Range to conserved grassland	177	657	–506	0	0.47
Range to conserved wetland	588	2,190	1,159	0	4.40
Range to conserved (average)	353	1,314	–253	7,013	2.16

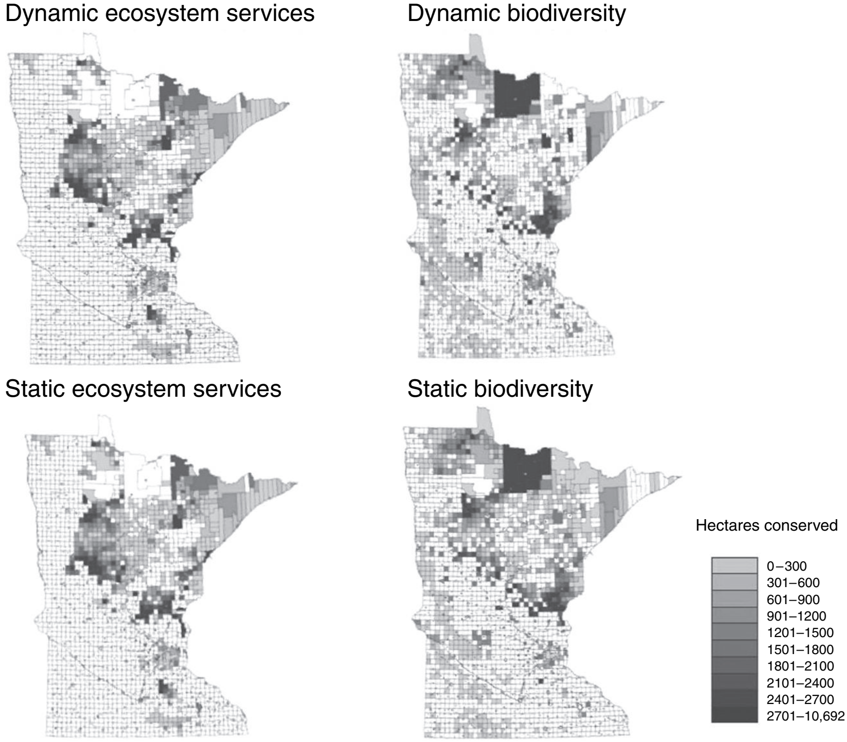

Fig. 13.1. Maps showing the location by sub-county unit of lands purchased for conservation under both ecosystem service and biodiversity conservation strategies for dynamic and static land-use scenarios

by using low and high carbon value scenarios and a high water-quality value scenario (Table 13.7). We did not use a lower value for water than the base case as this value was already fairly low. For the low value for the social cost of carbon we used a value of \$27.78 per ton of carbon (\$7.58 per ton CO_2) in 2011 dollars, which corresponds to Tol's value for the 33rd percentile from the fitted distribution, assuming a 3 per cent discount rate (Tol, 2009). For the high value for the social cost of carbon we used a value of \$240.32 per ton of carbon (\$65.54 per ton CO_2) in 2011 dollars, which corresponds to Tol's value for the 67th percentile from the fitted distribution, assuming a 0 per cent discount rate (Tol, 2009). For the high water-quality value we used a value from Mathews et al. (2002), who reported an average value of \$140 per household per year in 1997 dollars, or \$187.46 in 2011 constant dollars, for a 40 per cent reduction in phosphorus loadings in the Minnesota River. Though the value of ecosystem services changes dramatically with the large change in values, the strategies of what lands to choose and the impact on the

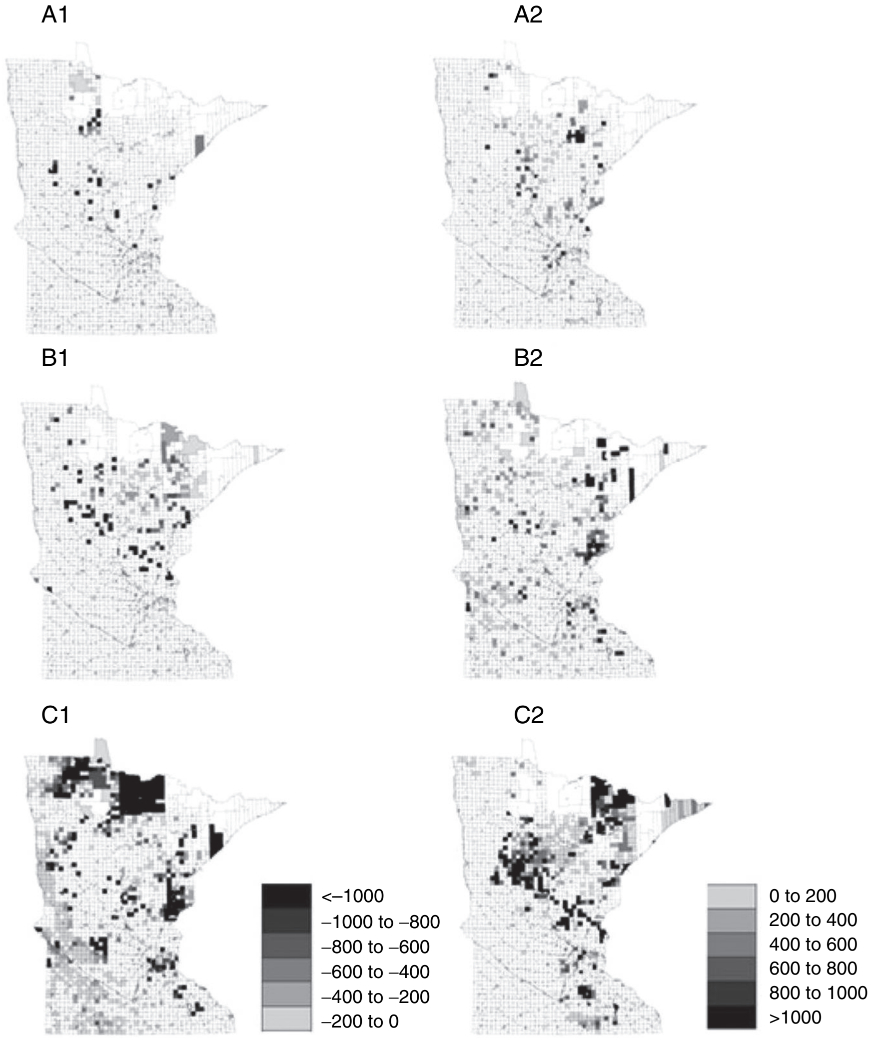

Fig. 13.2. The difference in lands purchased for conservation under the biodiversity conservation and ecosystem services objective across static and dynamic land-use scenarios

Notes: Sub-county units in the panels on the left indicate a decrease in the number of hectares conserved while the panels on the right indicate an increase in the number of hectares conserved. Panel A: number of hectares conserved for the ecosystem services objective under the dynamic land-use scenario minus the number of hectares conserved under the static land-use scenario. Panel B: number of hectares conserved for the biodiversity objective under the dynamic land-use scenario minus the number of hectares conserved under the static land-use scenario. Panel C: number of hectares conserved under the dynamic land-use scenario for the ecosystem service objective minus the number of hectares conserved under the dynamic land-use scenario for the biodiversity objective.

Table 13.7. Change in the value of ecosystem services (ES) and the biodiversity score between 2001 and 2026 for the State of Minnesota under various scenarios under low carbon, high carbon, and high water-quality value

Scenario		Private lands					Conserved land					Total (private + conserved land)	
		Biomass carbon ($m)	Soil carbon ($m)	Water ($m)	ES ($m)	Biodiversity score (m)	Biomass carbon ($m)	Soil carbon ($m)	Water ($m)	ES ($m)	Biodiversity score (m)	ES ($m)	Biodiversity score (m)
Dynamic conservation solution incorporating land-use change in planning													
ES objective: low carbon values	Mean	727	854	465	2,046	−8.24	1,585	147	752	2,484	4.9	4,530	−3.38
	SD	1.04	1.08	1.73	2.83	0.01						2.83	0.01
ES objective: high carbon values	Mean	6,215	7,370	501	14,085	−8.43	14,805	1,378	436	16,619	5.4	30,704	−3.06
	SD	10.24	8.13	2.19	16.24	0.01						16.24	0.01
ES objective: high water-quality value	Mean	3,345	3,892	2,822	10,059	−8.09	6,754	613	5,302	12,670	4.6	22,729	−3.51
	SD	4.71	4.70	10.35	14.92	0.01						14.92	0.01

Notes: All dollar figures are report in millions of 2011 constant dollars. Biodiversity scores are reported in millions of habitat units (representing predicted richness × habitat area).

biodiversity objective are relatively minor. The low carbon value and high water-quality value result in virtually the same overall biodiversity score, which is somewhat lower than the score for the baseline case (–3.38 and –3.51m versus –3.17m). These scenarios turn out to be quite similar because what matters in selecting lands to conserve is the relative weight between carbon and water. The areas important for water-quality improvement are those near population centres, where numerous households are affected by this. These areas also tend to have high land prices so that, overall, not as much land area is conserved when more attention is paid to water-quality improvements compared to carbon, which results in lower biodiversity improvement. Raising water-quality value or lowering carbon value each increase the importance of water quality relative to carbon. The high carbon value scenario does not make much change from the base-case scenario (–3.06m versus –3.17m), which already had most of the value of ecosystem services coming from carbon. Raising the carbon price skews the weight towards carbon even higher. With a low value for carbon and the base-case water-quality value, which also generated low water-quality values, we found that the increase in ecosystem service value with conservation was lower than the costs of conservation. On the other hand, for the high-value carbon case, we found a return on investment of over 4 to 1.

13.4 DISCUSSION

We find a high degree of alignment between strategies that target the value of ecosystem services and those that target habitat for biodiversity conservation. Targeting one of these two objectives generates 47–70 per cent of the maximum score of the other objective. In general, investing in conservation that increases the value of ecosystem services is also beneficial for biodiversity conservation, and vice versa. It is not surprising that there is good agreement between the outcomes of the two strategies, given the importance of biodiversity to maintaining the ecosystem function that supports the provision of ecosystem services. The choice of specific objective, however, does matter in terms of specific types of conservation investment to make. For ecosystem services under the base-case assumptions that place a relatively high weight on carbon sequestration, most conservation investments are made in the north-east and south-east portions of the state to maintain or restore forests. Little investment is made in the western portions of the state where the native habitat is grassland rather than forest. For biodiversity conservation, however, investments are made more evenly throughout the state to restore both forests and grasslands. Conservationists interested in either ecosystem services or biodiversity would do well to pay most attention to increasing the size of the conservation budget as the first-order objective. Increases in the budget will improve outcomes in terms of both objectives. The proper objective for conservation, biodiversity conservation for its own sake or increasing the value of ecosystem services, also matters and can shift the focus in terms of which particular areas are of highest priority for conservation.

We find that investing in conservation is highly beneficial. In the base-case analysis that includes the value of carbon sequestration and water-quality improvements, we find a return on investment of roughly \$2–3 per dollar invested. Only when we change the base-case assumptions to include a low value of carbon, along with the base-case value for water-quality improvement that is quite modest, do we find that the costs of conservation outweigh the benefits. Including higher values for carbon sequestration or water-quality improvement, or including a wider range of services, will increase the return on investment in conservation. As the data and ecosystem service models improve, it will be possible to move towards a more complete accounting of the values of conservation.

Biodiversity is a complex concept with multiple dimensions. There is great diversity in the published definitions of biodiversity and a wide variety of ways it can be measured (Mace et al., 2012). In this chapter we directed conservation funds towards one particular biodiversity conservation objective, namely the goal of conserving habitat for the benefit of vertebrate species. This objective reflects the Legacy Amendment's broad goal to protect game and wildlife species. Our biodiversity target considers the habitat requirements of 354 terrestrial vertebrate species. These species require a wide variety of areas and land-cover types and resulted in the selection of areas for conservation spread across the state (as shown in Figure 13.1). However, a broad-brush look at vertebrates could potentially mask more nuanced patterns and trade-offs. We would probably find different conservation strategies when targeting specific species or sets of species based on functional group, habitat preference, threatened status, charismatic species, or game species. On the other hand, using vertebrate species richness is a very limited measure of biodiversity if one takes the 1993 CBD definition: 'the variability among living organisms from all sources . . . this includes diversity within species, between species and of ecosystems' (CBD, 1993). Total biodiversity, including micro-organisms, primary producers, and a range of consumers including invertebrates and vertebrates, and the variability at the genetic and ecosystem levels, encompasses a wider set of biodiversity than we considered in this chapter.

Our habitat for biodiversity score captures the importance of habitat to support biodiversity but ignores several other factors. This measure does not account for the impact of habitat fragmentation or spatial pattern on species, which can be important for species with limited dispersal ability or in highly fragmented landscapes (Fahrig, 2003). It also assumes constant returns to scale in habitat provision. Polasky et al. (2008) use a more complex biodiversity model to estimate how land-use changes will affect species, accounting for fragmentation and variable marginal value depending on contribution of additional habitat for population viability. This approach, however, requires far more data and the use of sophisticated search algorithms for optimization, which makes its use impractical in many settings. Because each species is assumed to have equal intrinsic value, our measure gives equal weight to all species so that providing habitat for common species is of equal value to providing habitat for game species or threatened and endangered species. We ignore the different values to different species, namely the many use (e.g. game, pollination) and non-use values (e.g. existence, aesthetic) people derive from biodiversity. These additional values could be addressed by introducing species weights to reflect relative value, though it can be difficult to get agreement on the proper weights to use.

While we found that our broad measure of biodiversity was generally aligned with our measure of the value of ecosystem services (carbon sequestration and water quality), it is quite possible that other measures of biodiversity, such as those discussed in the prior paragraph, or other measures of ecosystem services might generate different results in terms of the degree of alignment between biodiversity conservation and ecosystem services (Bennett et al., 2009; McShane et al., 2011; Reyers et al., 2012). Mace et al. (2012) noted that there is a 'complex relationship' between biodiversity and ecosystem services. Biodiversity regulates ecological processes that support the provision of ecosystem services. In some instances, components of biodiversity contribute to the provision of services, as, for example, the contribution of genetic material to the discovery of new pharmaceuticals. In some cases, components of biodiversity are ecosystem services in their own right, as, for example, the cultural services generated by the existence or abundance of species. As regulators of ecosystem processes or as ecosystems services themselves, species and groups of species may be more closely aligned with services than is presented here.

Prior empirical analyses that examined different aspects of biodiversity and ecosystem services or looked at the impacts of particular decisions have found different degrees of alignment. For example, a large number of studies in ecology have examined the relationship between biodiversity (usually measured as plant species richness) and ecosystem functions and generally find increased diversity yields increased function (e.g. Loreau et al., 2001; Tilman et al., 2001; Balvanera et al., 2006; Isbell et al., 2011). Egoh et al. (2010), in a study in the Little Karroo in South Africa, found that meeting biodiversity conservation targets improved the provision of ecosystem services, but that for the same cost ecosystem services could be increased by far more if they were targeted instead of biodiversity. Overall, Egoh et al. (2010) found less congruence between biodiversity conservation and ecosystem services than we found in our analysis. Using data from the Willamette Basin in Oregon, Nelson et al.

(2008) showed that at-risk vertebrate species were maximized when conservation funds restored rare natural habitats, including oak savannah, prairie, and emergent marsh. Carbon sequestration, on the other hand, was maximized when conservation funds restored or conserved forests, including old growth, mixed, and riparian forest. Indeed, maximizing forest cover did benefit some species (e.g. the spotted owl); however, it provided little benefit for the majority of the thirty-seven rare species analysed. Nelson et al. (2009), also using data from the Willamette Basin, found that a conservation-oriented land-use scenario was better for biodiversity conservation and for non-market ecosystem services related to carbon sequestration, water quality (both reductions of phosphorus reduction and erosion), and reduction of flood risk, as compared with a business-as-usual and development-oriented land-use scenario. In Minnesota, Polasky et al. (2011) found trade-offs among different conservation strategies, particularly between species dependent upon different habitat types, grassland dependent birds, and forest dependent birds. However, they also found that strategies that ranked high in terms of the value of ecosystem services tend also to rank high for a general measure of biodiversity conservation. Several other studies in agricultural landscapes have found trade-offs between types of ecosystem services provided (e.g. provisioning services versus cultural and regulatory services) or between more intensive commodity production and biodiversity conservation (e.g. Santelmann et al., 2004; Boody et al., 2005). So, while we think that it will often be the case that what is good for promoting the supply of ecosystem services is good for biodiversity conservation and vice versa, one can always do better by targeting the objective of interest directly. Further, there is no guarantee that both objectives will always tend to be positively correlated, or that this will be true for particular components of biodiversity or particular ecosystem services.

Our analysis provides evidence on the degree of alignment between various conservation objectives, and it provides evidence on the net benefits of investing in conservation, but it is hardly the last word on either subject. In any type of integrated modelling such as this, there are always additional factors that can be considered. One important issue not considered here is land market feedbacks between conservation strategies and land prices (Armsworth et al., 2006), which then might drive land-use decisions on other un-conserved land. Land market feedbacks and indirect land-use change factor into the discussion of policies to reduce deforestation such as REDD (Reducing Emissions from Deforestation and Degradation; Miles and Kapos, 2008) and the impacts of biofuel expansion (Fargione et al., 2008; Searchinger et al., 2008). While we found positive return on investment for the level of investment in conservation under the Legacy Amendments, we did not attempt to solve for the optimal level of investment that would maximize social net benefits. Doing so would require building in price feedback effects which reflect relative scarcities that are a function of the land-use and management practices decisions made. Consideration of management practices, such as fertilizer application rates and tillage practices in agriculture, in addition to land-use change, can provide additional options that allow for improved performance on multiple dimensions. Finally, consideration of spatial interactions, where the benefit of taking action on one land parcel depends on what actions are taken nearby, and dynamic transition paths, such as the time path of accumulation of carbon with forest maturation rather than analysis of steady-state conditions, could provide additional insights. These would be interesting avenues to pursue in future work.

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14

Incentives, Private Ownership, and Biodiversity Conservation

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14.1 INTRODUCTION

Much biodiversity is found on privately owned land. For example, in the UK agricultural land provides important habitats for a wide range of birds and insects (UK NEA, 2011). In the US, at least one population of two-thirds of all species listed as being federally endangered is found on private land (Groves et al., 2000). Privately owned forest land in Finland and Poland contains many Natura 2000 sites,[1] a designation which is indicative of high conservation values (Watzold et al., 2010).

The way in which private lands are managed therefore has major implications for biodiversity. In Australia, conservation of many endangered native species depends on changing the behaviour of private farmland owners (Reeson et al., 2011),[2] while plant species richness in privately owned Austrian hay meadows has been shown to decline with increasing agricultural intensity (Zeckmeister et al., 2003). Changes in how agricultural land is managed have had significant impacts historically on a range of biodiversity indicators in the UK (Hanley et al., 2009), with the twentieth century, in particular, being associated with declines of many species groups on farmland (Wilson et al.,

[1] In May 1992 European Union governments adopted legislation designed to protect the most seriously threatened habitats and species across Europe. This legislation is called the Habitats Directive, and complements the Birds Directive adopted in 1979. At the heart of implementing these directives is the creation of a network of protected sites known as Natura 2000.

[2] George Wilson of the Fenner School, ANU has commented that '[i]n Australia most of the losses of wildlife have taken place on private land, because of agricultural development, land clearing and the impact of feral animals. This means that private landholders have a big role to play in getting them back. Landholders need an incentive to want to do that, and that incentive can be altruistic, or financial, or both.' (ANU website.)

2009). Agricultural land management continues to impact biodiversity. For example, according to the UK National Ecosystem Assessment (NEA), 'recent evidence suggests that about 67 per cent of 333 farmland species (broadleaved plants, butterflies, bumblebees, birds and mammals) were threatened by agricultural intensification in the year 2000' (UK NEA, 2011, p. 65). Globally, habitat destruction and degradation associated with conversion to agricultural land and intensification of agricultural land practices are leading drivers of losses of biodiversity and ecosystem services (MA, 2005).

These trends of biodiversity loss impose costs on society, since biodiversity plays a key role in sustaining the functioning of ecosystems, and thus in the provision of ecosystem services (Mace et al., 2012), while individuals have been shown in many studies to be willing to pay for biodiversity conservation (Kontoleon et al., 2007; Chapter 6 in this volume by Atkinson et al.). Yet the supply of biodiversity typically goes unrewarded by market forces owing to missing markets: private landowners usually receive no direct financial reward for enhancing or protecting biodiversity, owing to the non-rivalness and non-excludability of these benefits (Hanley et al., 2006). Indeed, protecting biodiversity typically comes at an opportunity cost to landowners—for example, if it requires forgoing profitable land conversion or intensification. The market thus generates too little biodiversity conservation effort, and too much biodiversity loss. For this reason, government intervention to promote biodiversity conservation on private land is warranted.

Owing to a political reluctance to force landowners to produce more biodiversity, and practical issues with extending the planning system to agricultural and forest land management, governments in many countries have introduced a range of schemes whereby landowners and managers can *voluntarily* opt to take up contracts for changing how they manage land in return for payments. In Europe, such schemes are known as 'agri-environment schemes', or AES. Spending on AES has been rising as a fraction of total public spending on agriculture in the EU and the US. The EU spends on average US$7.2 billion per year on payments to incentivize farmers to enhance environmental benefits, including biodiversity, and to avoid using environmentally detrimental production techniques (Cooper et al., 2009). Within the UK, the largest AES has funding of around £400 million per year over the period 2007–13 (Dunn, 2011). The largest scheme in the US, the Conservation Reserve Program, spends US$1.7 billion per year (USDA, 2010).

Our focus in this chapter is on the design of such agri-environment schemes. To illustrate, we draw on examples from Europe, Australia, and the US. However, it is also important to recognize that AES provide a useful template for informing the design of 'payments for ecosystem service' (PES) programmes more broadly, and many of the issues that we discuss have parallels in debates about designing PES programmes in many other countries (Jack et al., 2008; Quintero et al., 2009; Chen et al., 2010; OECD, 2010; Sommerville et al., 2010).

We first of all review the economic characteristics of the 'biodiversity policy design problem', before moving to consider a range of policy options, and a series of policy design challenges. The sensitivity of agri-environment schemes to changing market conditions is also explored. We close by offering a classification system by which most policy options for biodiversity conservation on private land can be described in terms of their most important features from an economics viewpoint.

14.2 THE ECONOMIC CHARACTERISTICS OF THE 'BIODIVERSITY PROBLEM'

The idea of paying farmers for the production of environmental 'goods' such as biodiversity or landscape quality as part of the Common Agricultural Policy of the European Union evolved in the 1980s. Prior to this, the CAP was focused very much on price support and income stabilization for farmers. However, the impacts of the system of guaranteed prices, import quotas, and export subsidies were a great intensification of agricultural production and expansion at the external margin, leading to considerable losses in semi-natural habitats and declines in farmland biodiversity (Bowers and Cheshire, 1983). Coupled with evidence of rising burdens on EU taxpayers (due to the public subsidy given to agriculture) and consumers (due to the consequences for consumer prices), this led to public pressure for the reform of the CAP in a way which would mitigate adverse pressures on the environment whilst reducing the monetary costs of over-production (Lowe et al., 1986; Allanson and Whitby, 1996). The UK government was the first to introduce payments for voluntary participation in pro-environment management schemes in the Norfolk Broads in the mid-1980s, funded from the agriculture ministry budget. The 1986 Agriculture Act allowed the creation of a national network of areas where farmers could be offered payments for conservation-friendly farming (the Environmentally Sensitive Areas scheme). Partial funding of national AES-type schemes from EU farm budgets was made possible in 1987. Finally, in 1992, the EU's 5th Action Programme on the Environment led to the setting-up of a framework which facilitated the creation of nationally designed AES across the entire EU, under the Agri-Environment Regulation 2078/92. This was accompanied by a significant reduction in payments for production and the introduction of set-aside schemes, as attempts to reduce the budgetary cost of the CAP in an expanding European Union (Hanley et al., 1999). Spending on AES has since grown significantly, both in absolute terms and as a percentage of the CAP budget. In the rest of this section, we review the main aspects of the problem of encouraging private landowners to supply more biodiversity, using AES-type schemes.

Landowners often face a cost in taking actions intended to produce biodiversity. This cost can be expected to vary, both across landowners, and for any landowner according to the 'amount' of biodiversity she/he aims to 'produce' (Armsworth et al., 2012). Variation in this supply price across landowners comes from variations in opportunity costs, which may be due in turn to differences in land productivity, differences in production opportunities, differences in resources, and differences in skills. For example, Hanley et al. (1998) found that the opportunity cost for farmers in the Shetland Isles of reducing grazing intensity to improve the ecological quality of moorland varied from £5.70 to £21.87 per sheep removed from grazing moorlands. AES in which payment rates do not vary across landowners will over-compensate all but the marginal farmer if the opportunity costs of taking actions intended to produce a given level of biodiversity improvement differ across farmers. A cost-effective distribution of biodiversity supply effort will involve either the targeting of actions at low-opportunity-cost sites (Ando et al., 1998), or the use of economic incentives which encourage low-cost suppliers to offer to supply biodiversity outputs, rather than high-cost suppliers (Connor et al., 2008).

Second, for a particular landowner, the marginal cost of taking actions intended to produce biodiversity may also be increasing. For example, a farmer will give up the least productive land first for a subsidized wetlands recreation scheme, before giving up more productive land. The same principle applies to choices individuals will make over the enterprise mix on the farm—lower-cost changes that are compatible with biodiversity improvements will be made first.

Third, the marginal benefits of actions in terms of biodiversity 'produced' may also vary across landowners and respond in non-linear ways to the actions of individual farmers. For example, assume that the action needed to increase abundance of a particular bird species is to reduce livestock grazing intensity, and that this is costly for the farmer. A given reduction in grazing intensity can produce varying responses in terms of bird abundance for reasons to do with the characteristics of an individual site (e.g. its soil type, altitude, or exposure), the characteristics of neighbouring areas (e.g. the presence of woodland within 100 metres), and current grazing intensities already present on the site (Dallimer et al., 2009). For species-protection programmes, actions by a given landowner—for example, in refraining from the felling of old-growth forest—may have marginal pay-offs in terms of species recovery which vary with distance to the nearest existing population of the species. This implies that an efficient policy design would have incentives which vary across space, since the biodiversity pay-off per euro also varies; and/or that the awarding of conservation contracts would partly depend on spatially varying ecological benefit functions (conservation metrics), such as are used in Australia for scoring bids (Oliver et al., 2005; Connor et al., 2008).

A fourth feature of the biodiversity problem with economic importance is that of hidden information (Moxey et al., 1999). This is of two types. First, a regulator will typically be unsure about the cost type of individual landowners in terms of their true marginal supply prices for biodiversity. We have already argued that variations in these supply prices across agents are to be expected. But this information is hard for the government to observe, since it depends on a wide range of landowner and land characteristics, and since there is typically a large number of farmers/landowners who are involved in the supply of biodiversity. Farmers will have private information on these supply prices—whether they are 'high-cost' or 'low-cost' type. Farmers also have local knowledge of their land, which means they may have more information than the regulator on the likely ecological outcomes of certain actions—for instance, if they know of the existence of bird populations on their land of which the regulator is unaware. Second, AES often involve land managers undertaking 'actions' which are hard for the government to monitor accurately. For example, if increasing populations of the bush stone-curlew in Australia requires farmers to engage in predator control, such actions are very hard (and costly) to monitor for the agency paying for these actions by way of conservation contracts. The level of effort which farmers engage in to fulfil the terms of their contracts is not known to the regulator with any precision. If this is so, then given that effort is costly to the farmer, farmers have an incentive to shirk and not undertake the actions for which they are being paid. This in turn means that the expected biodiversity benefits are not forthcoming.

Hidden information on farmers' cost type, ecological potential, and hidden actions leads to problems of adverse selection and moral hazard, the implications of which are usually analysed within a principal–agent model (Mueller, 1989; Fraser, 2002; Ozanne and White, 2008). Anthon et al. (2010) model the effects of these problems on the optimal design of incentive contracts for Natura 2000 forests. In their paper, ecological benefits from landowner actions are unknown before a conservation contract is signed, and only revealed *ex post*, and vary across forests. They show that the regulator should optimally offer forest owners an amount greater than their true supply price in order to induce compliance on high ecological-potential sites. They also conclude that payments should at least partly be linked to observable ecological outcomes, rather than just the cost of actions. However, most AES at present are based on actions, not outcomes.

A final feature of the biodiversity problem which is important for economic analysis is that the biodiversity benefits of a particular set of actions are stochastic from the viewpoint of the individual farmer/forest owner, since they are only partly a function of the actions of this agent. Consider the case of actions designed to increase the population of a bird species that nests and breeds on farmland. Ecologists know that certain actions that farmers can take are likely to contribute to an increase in the overall population size of this species. Such actions might include predator control, creation of small wetlands, and appropriate grassland management. But the abundance of the species on any one farm will be highly variable through time and will also respond to many factors outside the farmer's or regulator's control, including, for example, climatic variations, variations in the abundance of parasite species, variations in abundance of competing species, etc. This means that the outcome which the regulator cares about is only partly under the control of the agent charged with producing it. For risk-averse agents, this means that they face a cost of risk-bearing from non-delivery of the environmental good. This matters if an AES is set up to pay for biodiversity outcomes rather than actions (see section 14.4). In such circumstances, it may be necessary to offer farmers a two-part payment, one which depends on actions, and one which depends on outcomes. In this way, the government shares the cost of risk-bearing.

14.3 POLICY DESIGN OPTIONS

14.3.1 Regulation

Governments have the option of compelling landowners to protect biodiversity on their land—for example, by refraining from certain potentially damaging operations for specific sites or specific species. The former approach was followed in the Wildlife and Countryside Act in UK, while the latter is exemplified by the US Endangered Species Act (ESA). Two problems follow from such legislation. First, legislation often fails to recognize the (opportunity) costs which designation of protected species puts on landowners, and thus creates conflicts (Brown and Shogren, 1998). It also leads to incentives for landowners to take actions which downgrade sites so that they are de-listed, and thus controls removed. Thus a landowner in the US, finding a federally listed species on their land, has an incentive to destroy this species, and thus avoid the restrictions which its public discovery would place on them (Brown and Shogren, 1998). The US ESA has undergone various revisions in a bid to address some of the incentive problems created for private landowners, through, for example, the introduction of the 'no surprises' clause in habitat conservation plans, and the introduction of 'safe harbour agreements' (Bean, 2000). The UK Wildlife and Countryside Act recognized that costs would occur as a result of restrictions of 'potentially damaging operations' on sites of special scientific interest, and offered to pay compensation for profits forgone from such actions. But this led to an incentive for landowners to threaten to undertake such actions, since the only means the Nature Conservancy Council had of stopping them was to offer payments, leading to a problem of moral hazard (Spash and Simpson, 1994).

Moreover, the extension of detailed regulatory control over the actions of private landowners in the countryside with respect to agricultural and forest management has not found political favour in many Western countries, owing to the nature of *de jure* and *de facto* property rights over rural land use. Thus, extensions of the planning system (for example) to cover agricultural land use are uncommon,[3] and, indeed, might be very inefficient due to variations in supply prices for biodiversity across landowners, and the likely magnitude of the administration costs of enforcing such an extension of planning rules.

14.3.2 Uniform payment schemes

Uniform payment schemes dominate agri-environmental policy. Farmers are offered a payment for a set of management actions which are thought to increase biodiversity. In many cases, such payments are available only within certain geographic regions of a country; in others they are available countrywide. Uniform payments have a number of advantages. They are relatively simple to set up and to administer, and may be perceived as 'fair' since every landowner is offered the same price for undertaking a given action. Uniform payments are more cost-effective than regulation, since only those farmers with a supply price less than the subsidy will sign up. However, such schemes ignore many of the features of 'the economic problem', in that they over-reward all but the marginal producer; while usually no recognition is made of spatial variation in the supply price or variation along the supply curve for a given farmer. Payment rates may or may not recognize variations in ecological potential of sites, depending on the basis on which they are calculated—for example, a calculation of average opportunity costs of complying with a set of management measures would not reflect variations in ecological potential. An improvement would be to allow spatial targeting of payments across farms (Armsworth et al., 2012).

A variant on the model of offering payments to individual farmers for conservation actions is to offer payments for teams or groups of land managers to sign up, which can encourage spatial coordination of actions as well. This approach is epitomized in the Netherlands. Policy-makers there have developed schemes in which farmers work in collaboration with each other and with local, regional, and national agencies. By 2004, cooperative agreements existed between 10 per cent of all farmers in the Netherlands, covering 40 per cent of all agricultural land (Cooper et al., 2009). On a much smaller scale, the UK's Higher Level Stewardship Scheme offers a financial incentive for group applications for a single management option, although uptake seems to be rather limited (Franks, 2011).

[3] Although aspects of farmers' activities in UK national parks, for instance, may be regulated by planning procedures (e.g. the construction of new agricultural buildings).

14.3.3 Conservation auctions

Conservation auctions are reverse or procurement auctions, where the auctioneer—the policy-maker—procures environmental benefits such as biodiversity improvements from a selected set of landowners. These landowners are chosen on the basis of their submitted bids which reflect their supply price. These bids are anchored from below by the opportunity costs of changing land-use management and may have institutionally fixed upper limits or 'bid caps'. An auction fosters competition between bidders to minimize the 'information rents' or profits earned by landowners, and maximize the amount of ecosystem services procured for a given budget, since lower bids have more chance of being accepted. Given a fixed budget for contracts, farmers have an incentive to moderate bids if they wish to be awarded such an agreement (Stoneham et al., 2003; Rolfe et al., 2009).

Perhaps the most prominent conservation auction is the Conservation Reserve Program (CRP), which was started in 1985 by the US Department of Agriculture (USDA, 2011). Under the CRP landowners' bids are ranked in descending order on the basis of a benefit–cost index, termed the Environmental Benefit Index. The benefit element of the ratio is the ecological value of the environmental benefits supplied by the project, and the cost is the bid submitted (monetary values for these benefits are not computed). Use of this benefit–cost ratio discourages landowners from marking up their bids too high as this reduces chances of selection. A range of auction mechanisms, such as the BushTender (Stoneham et al., 2003), Catchment Care Australia (Connor et al., 2008), and the Auction for Landscape Recovery pilot (Gole et al., 2005), have been employed in Australia. Brown et al. (2011) describe an auction in the Canadian Prairies linked to conservation easement payments. Schilizzi and Latacz-Lohmann (2007) provide a critique of early findings on the cost-effectiveness of actual conservation auctions.

Several design options exist for conservation auctions (Schilizzi and Latacz-Lohmann, 2007). For example, a government needs to decide whether to use a uniform price design (all successful bidders receive the same payment) or a discriminating price design (successful bidders receive their bid price). Uniform price designs can do a better job of revealing true opportunity costs, since if the price is set equal to the highest losing bid, then an individual farmer's bid determines only the chances of winning a contract, not the value of the contract. However, uniform price designs may deter participation (Brown et al., 2011). In multi-round iterative auctions, participants can submit bids repeatedly in multiple rounds. In these auctions, bidders get the opportunity to revise their bids. Thus losing bidders have a chance of lowering their bids and getting accepted in latter rounds. Schilizzi and Latacz-Lohmann (2007), Cason et al. (2003), Cason and Gangadharan (2004), and Rolfe et al. (2009) indicate that inter-temporal learning in general reduces the cost efficiency of the auctions relative to a subsidy, irrespective of the ecological goal, or that there is only a very modest improvement of performance over time (Cason and Gangadharan, 2005).

As noted, several studies have indicated that inter-temporal learning can negate the efficiency gains promised by conservation auctions over uniform payment schemes. However, there are many ways to design conservation auctions, and field trials or economic experiments necessarily examine a small sample of possible configurations. To overcome such limitations, a number of authors have turned to simulations. For example, Lennox and Armsworth (2013) used agent-based modelling to test the performance of various multi-round conservation auctions. As the auction rounds progressed, landowners were able to learn and adjust their bids in an attempt to maximize their surplus. A central feature of the authors' analyses was the role of landowner cooperatives. Specifically, do landowner cooperatives increase the ability of landowners to gain surplus from conservation auctions?

The results show that auction performance depends on a complex interaction between the objective of the conservation agency, the extent of landowner cooperation, and the level of the bid cap (Figure 14.1). Auctions in which conservation benefits are complementary, such as those focused on conserving overall species richness, result in the potential for landowner cooperatives to gain large amounts of surplus, creating an incentive for cooperative formation (Figure 14.1*a*). In contrast, those auctions in which conservation benefits are substitutes, such as those with the objective of maximizing the number of land parcels enrolled, cede limited surplus to landowners and create a disincentive for the formation of cooperatives (Figure 14.1*c*). Placing a low cap on landowners' bids checks the ability of cooperating landowners to gain large surpluses in those auctions with complementary benefits (Figure 14.1*a*). This comes at the expense of conservation outcomes, however, which can be significantly diminished in comparison with those auctions that place no cap on landowner bids (Figures 14.1*b* and 14.1*d*). Conservation agencies focused on conserving species richness, which value biodiversity most directly, therefore face a critical trade-off in employing conservation auctions. For auctions to be ecologically effective they must be structured such that they harbour the potential to cede considerable surplus to landowners. Minimizing landowner surplus, on the other hand, may lead to the conservation of less biodiversity than would otherwise be the case.

The relative effectiveness of different conservation auction designs in any context also hinges on political, economic, and ecological considerations. Connor et al. (2008) note that assessments of the performance of auctions relative to uniform payments can be undertaken by fixing either the total cost of the scheme, or the environmental success (e.g. acres enrolled), with rather different conclusions emerging about comparative performance. Moreover, changes in the design of auctions could have differing impacts on alternative

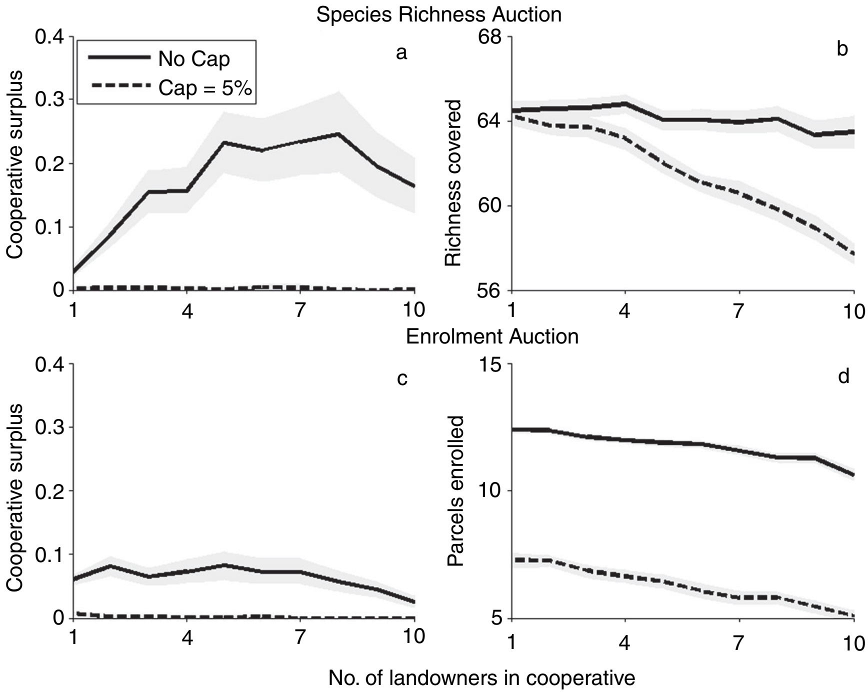

Fig. 14.1. Surplus obtained by landowner cooperatives and conservation outcomes

The surplus obtained by landowner cooperatives (*a* and *c*) and conservation outcomes (*b* and *d*) after the tenth round of conservation auctions when the objective is to maximize species richness over selected sites (*a* and *b*), and when the objective is to maximize the number of land parcels enrolled (*c* and *d*). Surplus is expressed relative to the cost of the selected sites if the landowners of those sites had received no surplus. Figures presented are the mean after 100 simulations along with 95 per cent confidence bands.

Source: Lennox and Armsworth (2013).

criteria of policy performance. For instance, increasing information about the spatial location and characteristics of other bidders might improve ecological outcomes at the expense of higher pay-outs to farmers (Cason et al., 2003). Finally, the design of the conservation metric with which bids are weighted is crucial to determining the success of auctions (Connor et al., 2008).

14.3.4 Conservation easements

Conservation easements provide a popular policy option for securing conservation improvements on private land in many parts of the world (Environmental Law Institute, 2003; Merenlender et al., 2004; Land Trust Alliance, 2011). Easements are voluntary, market-based agreements between landowners and conservation agencies in which the landowner receives a direct payment and/or tax rebate in recompense for ceding particular land rights. However, the landowner retains overall fee title to the property. Some easements have been used to limit development, but many others place restrictions on grazing activities, timber operations, etc. Moreover, while a popular tool for land trusts and other non-profit organizations, easements are also commonly used by government agencies to secure conservation gains. Taken together, these aspects make easements often very comparable to AES. Unsurprisingly then, when designing easements, policy-makers face many of the same challenges that are present in designing AES. For example, hidden information about the true supply price of conservation benefits on a given property make it challenging for regulators to avoid overcompensating all but the marginal landowner (Armsworth and Sanchirico, 2008). Similarly, government agencies face challenges in monitoring compliance with easement terms.

One obvious difference between easements that restrict agricultural uses and timber extraction on a property and AES is that the exchange of property rights in an easement is commonly made 'in perpetuity', whereas AES typically offer fixed-duration contracts. In general, the advantages and disadvantages of operating conservation contracts of different durations are not a well-studied area, something that we return to shortly. Also, often large numbers of AES contracts are being issued simultaneously in scheduled (re-)enrolment rounds. In contrast, easement transactions often occur in a more piecemeal fashion, proceeding on a deal-by-deal basis, which limits scope for relying on competitive allocation mechanisms to overcome limitations of hidden information. Despite these differences in how the two instruments are being applied, we believe that much could be learned from comparative studies contrasting experiences with AES and easements.

14.3.5 Creating markets for biodiversity

One aspect of the 'biodiversity problem' outlined in section 14.2 is that of missing markets. Since many of the benefits which biodiversity conservation provides are non-rival and non-excludable, markets may not emerge in which buyers and sellers trade. However, government agencies sometimes enable such markets to form. For example, the US Fish and Wildlife Service has sometimes allowed trading in endangered species and their habitats under the US ESA (Bean and Dwyer, 2000; Fox and Nino-Murcia, 2005) following a cap-and-trade-type approach. Under this model, a landowner who plans to undertake land management actions that may harm individuals of a federally listed species is required to undertake compensatory mitigation to improve the plight of the species elsewhere. This could involve purchasing species conservation credits from a third-party mitigation bank that specializes in creating and restoring habitat for the species on a different site. The potential economic benefits from such a scheme are realized through the gains from trade made possible by introducing flexibility into the command-and-control regulation. Ecological benefits could also result by allowing otherwise disparate conservation actions on the landscape to be aggregated in space. Also, some species require proactive management of habitats, such as fire management, something that can be incentivized with this approach but otherwise is not covered by the US ESA. As originally framed, the US ESA prohibited private landowners from taking certain actions that would harm listed species, but did not require them to undertake conservation management that would aid these species. Despite the proposed benefits of such trading schemes, it should be emphasized that designing and implementing conservation banking programmes in such a way that promises that economic and ecological benefits are realized is a formidable policy challenge in its own right (Salzman and Ruhl, 2000).

Markets for biodiversity can also arise in the absence of a regulatory cap—for instance, if private buyers can capture some of the benefits of conservation. Conservation organizations can offer conservation contracts to farmland owners, with their members benefiting from resultant conservation outcomes (more birds), an example being the Ducks Unlimited Canada scheme in prairie habitats of Alberta, Manitoba, and Saskatchewan, which offers payments to farmers for wildfowl-friendly farming practices (Banack and Hvenegaard, 2010). Numerous voluntary markets are also starting to emerge where buyers pay for the delivery of specified ecosystem services—for instance, water companies paying farmers to reduce run-off of water pollutants by changing how they manage livestock, or paying landowners for peatland restoration as a way of reducing downstream water treatment costs (Dunn, 2011). Such voluntary markets are much less abundant for biodiversity conservation, presumably because the private benefits of increases in biodiversity are lower and dispersed across many more beneficiaries than, say, the increase in profits to a single water company from a reduction in water treatment costs. Biodiversity improvements could be purchased by bundling them with water quality improvements. Government's role in such emerging markets may be as a facilitator, as in the setting of codes of practice (although these can also emerge from the sector without intervention), and as a regulator of trades.

All of the policy options discussed in this section are based on the notion that landowners are primarily motivated by profit maximization, and that monetary incentives are required to encourage them to supply costly biodiversity benefits. In a competitive industry (such as farming), where the great majority of producers are price-takers and sell undifferentiated products, profit maximization is the strategy most likely to be consistent with long-term economic viability. We thus think profit maximization is a reasonable assumption to make for the representative farmer's motivation. This is not to dispute that other motivations are important, as summarized in the recent paper by Sheeder and Lynne (2011). They ask whether profit-maximizing is a reasonable assumption in describing the environmental behaviour of farmers when deciding whether to take up AES payments. They give examples of empirical studies showing that the assumption is reasonable—for example, for conservation auctions in Australia and participation in soil conservation programmes in Maryland, but also of cases where the assumption did not predict well (e.g. Chouinard et al. (2008) in the Pacific North-west). We also note that several schemes operate on the basis that farmers can be persuaded to adopt conservation-friendly behaviour if simply provided with information on, for instance, grassland management techniques which promote the survival of ground-nesting birds (Beedell and Rehman, 1999, 2000). If farmers are, indeed, willing to engage in conservation-friendly action voluntarily, then it is possible that offering monetary incentives may crowd out behaviour which is so motivated.

There are thus many options for policy-makers to choose from which are capable of increasing the supply of biodiversity from private land. These range from regulation, to offering fixed-price incentives, to setting up institutions in which landowners and managers can bid for contracts to engage in biodiversity conservation, to the creation of markets where buyers and sellers of biodiversity interact with each other. Each option varies in its ability to deal with the set of problems we identified in section 14.2, and we might expect a high degree of variation in the economic and ecological efficiency of these options. In the next section, a range of challenges which face policy-makers wishing to introduce these options is reviewed.

14.4 POLICY DESIGN CHALLENGES

In this section we identify some challenges for conservation policy design.

14.4.1 Paying for outcomes not actions

Since the objective of biodiversity policy is to increase the supply of biodiversity, an obvious question is whether payments should be targeted at outcomes (more bird species, higher species density) rather than at the management actions thought to lead to such outcomes. Most agri-environmental policy is, indeed, targeted at management actions, typically because these are thought to be easier to observe, and because the 'output' of biodiversity from a given area of land is determined by a wide range of factors, only some of which are under the control of the landowner. This means that outcome-based contracts are riskier for the landowner than action-based contracts (Whitten et al., 2007). Moreover, it may be more expensive for the regulator to monitor conservation outcomes (e.g. counting birds) than management actions (e.g. whether a farmer has drained a wetland or not). However, outcome-based payments have other advantages (Gibbons et al., 2011). If some of the management actions which are crucial to achieving a biodiversity target are hidden (very expensive for the government to observe), then paying for outcomes may be more efficient. Moreover, landowners and managers are quite likely hold information on the best areas of land within their properties for promoting target species populations, and may have alternative options for encouraging such increases in species. Outcome-based payments encourage land managers to make use of this information to generate biodiversity conservation more efficiently than payment for actions.

Whitten et al. (2007) consider the case of promoting conservation of ground-nesting birds in the Murray Catchment in Australia. From the perspective of the regulator, enhancing populations of birds such as the bush stone-curlew and brolga requires a combination of observable actions (e.g. stocking levels) and hard-to-observe actions, such as predator control and the day-to-day movement of stock. Moreover, landowners are likely to have private information on where on their land it is best to promote population increases of these birds. The authors present a theoretical model which combines an auctioned up-front payment for management actions with an *ex post* payment for conservation outcomes. They find that setting the outcome payment relatively high compared with the up-front payment is desirable, since it induces landowners with high ecological potential to enrol and to supply higher levels of conservation effort, although this is at the expense of fewer participants for a fixed budget. Whitten et al. then run a trial of the combined scheme with farmers in the area. Seventeen farmers made bids for contracts, with outcome-based contracts being preferred to action-based contracts. The costs of securing a given area of land enrolled were lower with outcome-based contacts, with a cost saving of around 30 per cent. Crucially, the researchers had developed a metric for measuring conservation outcomes in a relatively low-cost manner. In a similar vein, White and Sadler (2011) use a simulation modelling approach to investigate the design of a payment-for-outcomes scheme for native vegetation conservation in south-west Australia, which is based on an observable species metric for outcomes along with observable conservation actions (fencing) which can also be rewarded.

14.4.2 Determining contract length and other dynamic considerations

In AES, contracts with landowners are generally finite but span a variety of durations across different programmes (Lennox and Armsworth, 2011). Perpetual easements covering agricultural land uses can to some degree be thought of as an extreme case. Contract duration discussions are particularly salient given that ecological and economic conditions relevant to AES design vary through time, and future predictions about these conditions are subject to considerable uncertainty. Contract expiry can result in the loss of some or all of the ecological benefits supplied during the lifetime of the contract (Whitby, 2000).

Several theoretical studies are relevant to discussions of contract duration. Ando and Chen (2011) investigated the optimal length of conservation contracts in an analysis that incorporates enrolment and re-enrolment issues. They found that while longer contracts increase conservation benefits from any single landowner, they lead to fewer landowners being willing to re/enrol in the programme. The authors also show that contracts should be longer when the ecological benefits mature slowly, and that it may be optimal not to contract at all when uncertainty surrounds likely ecological outcomes. Finally, the authors show that non-ecological characteristics are also central to optimal length of a conservation contract; optimal contracts are longer where the turnover rate of parcels enrolled in conservation programmes is high and where the average private land income is low. Lennox and Armsworth (2011) also investigated how uncertainty regarding future ecological benefits of contracts and regarding a landowner's willingness to re-enrol on contract completion interact to determine optimal contract lengths. They find that uncertainty over future re-enrolment exerts more influence on the optimal choice of contract duration, and they also emphasize conditions under which a portfolio of contract lengths can outperform employing uniform-length contracts. Finally, in related work, Gulati and Vercammen (2006) examine a different dynamic aspect of conservation contracting and consider the potential benefits of offering time-varying payment schedules to recognize the changing incentive faced by landowners as a contract progresses and ecological conditions on the property improve.

14.4.3 Spatial coordination

Some elements of biodiversity (e.g. species with home ranges spanning multiple properties) can be more efficiently conserved if protection is targeted towards spatially adjacent parcels. Conservation agencies are thus sometimes interested in concentrating similar land uses on spatially connected parcels rather than dispersing them at different locations on the landscape. Conservation policies intended to achieve this spatial coordination have focused on combining uniform subsidy payments, which pay for the land-use changes, with top-up bonuses when neighbouring participants have similar land uses or have connections between patches which contain biodiversity-friendly habitats. Examples of such subsidies include those under the Conservation Reserve Enhancement Program (CREP) in the state of Oregon in the US, and subsidies with network bonuses in Switzerland (Mann, 2010). These policies have their economic foundations in the Agglomeration Bonus (AB), proposed by Parkhurst et al. (2002), Parkhurst and Shogren (2007), and Warziniack et al. (2007), which can incentivize spatial coordination. Communication between neighbours can produce the ecologically desirable outcomes, but may also imply a lower level of cost-effectiveness. Banerjee et al. (2012*b*) examine the effects of local network size on spatial coordination with an AB. They find that the strategic uncertainty of the AB coordination game environment, and the anonymity imposed by the network structure where only land-use choices of neighbours are visible to every player, produce significantly different spatial patterns of land uses. Additionally, coordinated areas of multiple land uses are also created on both networks. This result suggests that on real landscapes which closely resemble networks, we are most likely to see a combination of ecosystem services being delivered through coordinated management of land parcels incentivized by AB mechanisms.

Given these issues, attention has been devoted to implementing spatially connected auctions which give greater weight to bids that are spatially adjacent to each other. Reeson et al. (2011) and Windle et al. (2009) have experimented with such auctions, where spatial connectivity is one metric used to rank bids. A challenging proposition in the domain of spatial conservation auctions is to reduce intensified rent-seeking by participants at strategic locations on the landscape. As budgets are limited, if players at strategic positions exploit their locational advantage and submit very high bids, then too few projects may be procured and spatial patterns may not be attained at all. Thus the auction achieves neither economic efficiency nor ecological effectiveness.

14.4.4 Transactions costs

Transactions costs faced by landowners seeking to enrol in AES have been found to deter participation. The transactions costs incurred by participants can be classified into search, negotiation, administrative, monitoring, and enforcement costs (Dahlman, 1979; Hobbs, 2004). Of these, search, negotiation, and administrative costs are *ex ante* costs incurred prior to participation (Mettepenningen et al., 2009). The magnitude of these costs can play an important role in influencing farmer participation. McCann and Easter (1999) and Mettepenningen et al. (2009, 2011) estimate the transactions costs for water-pollution-reducing programmes in the Minnesota River in the US and for farmers and public agencies for AES participation in different parts of the EU. A study on AES participation in the EU highlights reduced participation of farmers in Sweden and Germany owing to such costs (Falconer, 2000). Moreover, complex conservation contracts with complicated ecological goals also increase transactions costs and discourage participation (Ollikainen et al., 2008). *Ex ante* costs, such as costs of filling in forms, going to workshops, negotiation, and joint planning between neighbours, are germane to the evaluation of the AB. Parkhurst and Shogren (2007) have analysed spatial coordination of neighbours as a coordination game. In their study, non-participation is a strictly dominated strategy since the pay-offs from the AB scheme are greater than the pay-offs from business-as-usual agricultural land use. This scenario may, however, change in the presence of transactions costs of participation. Agglomeration pay-offs can be obtained if neighbours participate and choose the same action as the player. Yet if the transaction costs of participation are high enough, eligible participants may opt not to participate at all. Additionally, if farmers reason that owing to high transactions costs, their neighbours will not participate, they may not participate either.

Banerjee et al. (2012*a*) consider the provision of two types of ES—one delivered via habitat connectivity and the other delivered through similar land uses within a particular distance of each other. The former ES is assumed to have a higher priority for the regulator relative to the latter one so that the AB payments associated with them are different. In the absence of transactions costs, for a given menu of AB payments, the coordination game has two Pareto-ranked Nash equilibria, with non-participation being strictly dominated. However, when considering the presence of transactions costs, the nature of the game changes. Since participation is voluntary, and in the absence of such costs is not necessarily dominated, we are faced with a dynamic game of imperfect information. For low value of transactions costs, there are two sub-game perfect Nash equilibria associated with the provision of both the ES. Yet with increasing values of transactions costs, the nature of outcomes may change with the two sub-game perfect Nash equilibria corresponding to non-participation, and with the delivery of ES relying on the creation of connected habitats across multiple properties. This result implies that when AES produce situations characterized by multiple Nash equilibrium outcomes, consideration of transactions costs can eliminate some of the inefficient equilibria (here associated with the distance-dependent land-use changes). This result suggests the need for experimentation in controlled lab experiments with student subjects, and in the field with actual landowners, to identify the scenarios under which the participation- and connectivity-based ES outcome can be obtained.

These four challenges (paying for outcomes or actions; determining the length of contracts; addressing the need for spatial coordination of actions across landowners; and the transactions costs of implementing a policy) require those charged with designing incentives for the promotion of biodiversity conservation on private land to weigh up what matters most, since there will often be trade-offs in the resolution of these challenges. For instance, encouraging greater spatial coordination through spatially coordinated auctions or an AB will make policy design more complex, which might lead to a rise in transactions costs. Similarly, paying for outcomes rather than actions might increase monitoring costs, depending on how outcome data is collected.

However, as Armsworth et al. (2012) show, it may still be beneficial to incur higher transactions costs through more sophisticated policy design, since this can improve overall economic efficiency and deliver more biodiversity benefits for a given overall social cost.

14.5 SENSITIVITIES TO MARKET CONDITIONS

The effectiveness of any given set of incentives for biodiversity conservation on private land will clearly depend on what other financial signals a landowner is faced with, along with the nature of the institutional structure within which they are operating, and such social and moral conventions and pressures which may also affect their decision-making. For example, if payments are offered to farmers to reduce stocking densities on grazing land in order to benefit bird communities, then the likelihood of farmers signing up for such contracts will likely depend on the price of outputs (livestock) which the farmer expects to receive, and also on the costs of producing these outputs (e.g. fertilizer costs). Institutional factors such as the existence of lump-sum subsidies for farmers or cross-compliance restrictions on qualifying for output subsidies may also matter. Different biodiversity indicators may be impacted to varying degrees by changes in output and input prices, depending on the 'production function' which links land management to species abundance or distribution.

Hanley et al. (2012) use a combined economic-ecological model of land use in the UK uplands to investigate this issue. The market condition parameters they make use of comprise meat and milk prices, and wages and fertilizer costs. Four future 'scenarios' of change in these parameters are used, based on a UK government 'Foresight' exercise. These scenarios are termed World Market, Global Sustainability, National Enterprise, and Local Stewardship. Each scenario also includes an assumption about the level of lump-sum subsidy available for farmers (the EU Single Farm Payment), production support levels, and agri-environment payments. Each scenario is then compared with a baseline equal to present-day conditions. Implications of these sets of incentives for land use are then modelled, and the resultant land management pattern is used to produce predictions of changes in:

- Densities of five different farmland bird species (curlew, lapwing, skylark, song thrush, and linnet);
- Total farmland bird density; and
- Total farmland bird species richness.

Of particular interest is the comparison of present-day baseline conditions with (i) the Global Sustainability scenario, as AES payments are held constant whilst input costs rise, meat prices fall, and milk prices rise; (ii) the World Market and National Enterprise scenarios, where AES payments are removed; and (iii) the Local Stewardship scenario, where AES payments rise by 20 per cent. Results vary according to the type of farm studied, but for a particularly common type (moorland sheep and beef), impacts on individual bird species are shown in Figure 14.2. The main message here is that winners and losers emerge from these changes in market conditions, dependent on how land management responds to changes in subsidy, input and output prices, and how individual bird species respond to these land management changes. Comparing the baseline present day with Global Sustainability, it can be seen that lapwing numbers fall whilst skylark and curlew increase. In the World Markets scenario, there are relatively large falls in lapwing but relatively large increases in skylark. Whilst it is hard to make general statements about the market parameters which are most important in determining the responsiveness of a biodiversity indicator to a given set of AES initiatives (since the use of the particular scenarios employed here means that, *ceteris paribus*, comparisons are not being made), the results do illustrate the complexity involved in relating changes in market conditions to the biodiversity outcomes of a PES-type scheme.

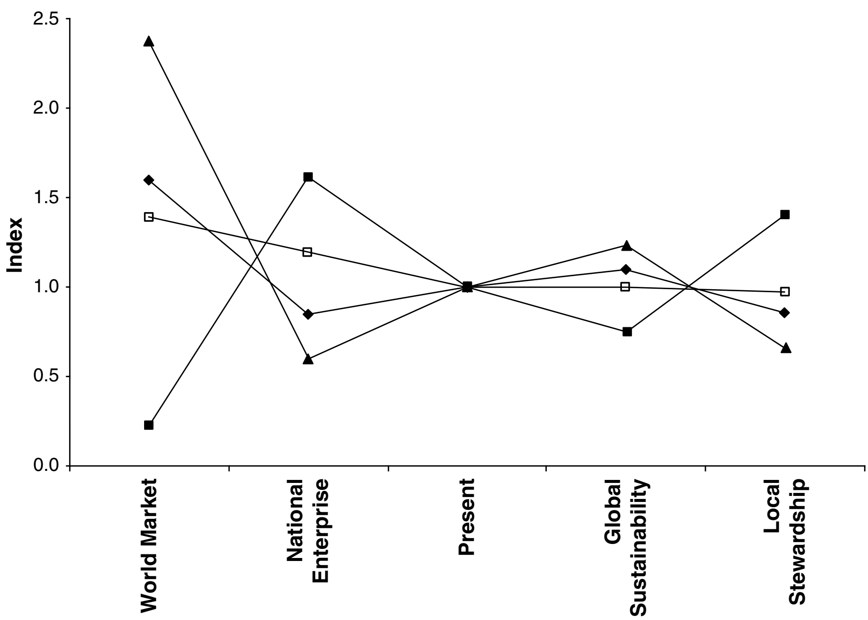

Fig. 14.2. Relative change in density of four bird species on moorland sheep and beef farms under Foresight scenarios

Key: Filled triangle: skylark; filled diamond: curlew; filled square: lapwing; open square: linnet.
Source: Hanley et al. (2012).

14.6 CONCLUSIONS

Designing better incentives to deliver more biodiversity conservation from private land is an important task. From the material set out in this chapter, it is possible to distinguish a set of policy attributes which are important. These are summarized in Table 14.1. Since the government is seen as contracting with private landowners for the supply of environmental goods, we describe these in terms of the nature of the contract arrived at. These attributes are: (i) the allocation mechanism—who the potential suppliers of the good are, and how they will be chosen; (ii) contract stipulation—what is to be supplied (e.g. hectares of wetland restored; reductions in stocking rates; or an output measure such as a density increase in a species of conservation concern); (iii) contract duration—how long the contract is for; and (iv) price—what payment is offered to the farmer, and how this is determined. We also highlight two possible responsibilities for fulfilling each of these aspects of mechanism design, according to whether responsibility lies with the principal—the government or its regulatory bodies—or the agents, namely the farmers or landowners. Thus in entry-level Environmental Stewardship, a government agency sets payment levels, but the overall allocation of contracts arrived at is determined entirely by which farmers choose to enrol. In contrast, with a conservation auction, farmers take responsibility for deciding what price they will receive for their actions when formulating their bids, and the principal chooses among bids to determine which of those contracts to accept. The more responsibility that the principal takes on for setting these design parameters, then the greater the burden of information acquisition—for

Table 14.1. Schematic of policy design attributes and responsibility for actions

	Principal decides	**Agent decides**
Allocation mechanism: who receives contracts	Conservation auctions, targeted enrolment programmes (higher-level ES in the UK)	Open enrolment programmes (e.g. entry-level ES in the UK)
Contract stipulation: what they must provide	For example, nitrate-sensitive areas in the UK	Higher-level ES in the UK where farmers choose what to offer from a menu of options
Contract duration	CRP in the US, ES in the UK, etc.	
Price	Entry-level ES in the UK. Higher-level ES pays a fixed price per contract, but farmers may offer differing numbers of actions	Reverse auctions such as CRP in the US, and numerous conservation auction programmes in Australia

Notes: ES = Environmental Stewardship; CRP = Conservation Reserve Programme.

instance, on farmers' costs and ecological benefits—it must bear. This framework also highlights one obvious gap in the current panoply of programme designs: namely, a scheme in which farmers compete in part by offering to commit to contracts of varying durations. Finally, we note that an important feature of all schemes is the state of knowledge about the ecological production function linking landowner actions to biodiversity outcomes; landowners and the government may know different things about such functions.

The policy options considered here are frequently implemented in a decidedly second-best world. Agricultural activity continues to be heavily subsidized in many countries, and such subsidies have in the past been argued to result in an intensification of production and an expansion of the area of land under farming which resulted in species declines for fauna and flora (Bowers and Cheshire, 1983; see also references in Dallimer et al., 2009). Reducing or removing agricultural subsidies might thus result in an improvement in the conservation status of many farmland species, although such impacts are not likely to be uniform in direction or extent across species, as noted in section 14.5. Farming also results in a range of negative externalities such as non-point nutrient pollution, which can reduce aquatic biodiversity (Dodds et al., 2009). Correcting such negative externalities should also be part of the portfolio of policies considered.

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On the Potential for Speculation to Threaten Biodiversity Loss

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15.1 INTRODUCTION AND MOTIVATION

A number of key wildlife species are threatened with extinction because of over-harvesting, habitat destruction, invasive species, pollution, or a combination of these factors. In recent years, analysts have become interested in the interaction between conservation of *in situ* wildlife species and the existence of *ex situ* stocks of wildlife commodities. Writing about basking sharks, for example, *The Economist* (2002) describes a shark-fin trader who:

> is so convinced that stocks are collapsing that a few years ago he cornered the market in Norwegian shark fins and stockpiled the result in Japan. He still seems confident that his stockpile will make him a fortune. (p. 85)

Meecham (1997) describes an encounter with a Japanese gentleman:

> who is breeding a pure strain of Hokkaido brown bear taken from the wild . . . He talks with pride about how he will have the one and only last pure strain of Hokkaido brown bear . . . His investment pays off big time. (p. 134)

Often, products from such species are believed to have important medicinal value (tiger bones, bear bladders, rhino horn), explaining why prices can increase substantially when supply is restricted, and why gaining market power is profitable for private investors. In other instances, consuming products from these species is seen to convey social status (shark-fin soup), and scarcity may induce 'Veblen effects', whereby preference for a good increases with price due to perceived exclusivity (Leibenstein, 1950).

Under certain conditions, it may be profitable for a speculator to *actively contribute* to the depletion of common stocks, speeding up or indeed triggering the extinction process. Anecdotal evidence supports this point: Meecham (1997), again, writes that:

> [m]assive stockpiles of rhino horn have been discovered, along with anecdotal reports from poachers claiming to have been instructed to kill rhinos in the wild whether they have usable horns or not. If the animal becomes extinct . . . those stockpiles become infinitely valuable. (p. 134)

Similarly, Kremer and Morcom (2000), citing anecdotal evidence in the *New York Times*, suggest that large-scale killing of wild rhinos—even dehorned ones—increases the value of *ex situ* stocks (p. 231). The Atlantic blue fin tuna provides a more recent example. One firm (the Mitsubishi Corporation) controls a large share of the market, and there has been speculation that these tuna are being frozen in anticipation of future price rises.[1] Concerns about potential extinction have led to suggestions that trade in tuna be banned under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), a move which Japan (the world's largest consuming country) has indicated it would ignore. Evidently, stockpiling tuna may well pay substantial dividends.

The economic forces underlying these examples are simple enough. As species become rarer, supplies from the wilds will dwindle and prices will increase, inviting additional pressure on extant populations. In other words, extinction may be an incentive-driven process, via the price mechanism.[2] These forces are exacerbated when a market player holds significant stockpiles of wildlife commodities. While these factors are undoubtedly of some importance, we believe another factor is potentially important—strategic behaviour. Certain agents may have an incentive to drive species to oblivion, and 'bank on extinction'. We define 'banking on extinction' as the behaviour of private parties investing in private stores of renewable resources (including endangered species), hoping that the combination of ineffective conservation efforts and high prices on consumer markets will deplete *in situ* stocks in the immediate future. With common stocks depleted, such investors may enjoy considerable market power and, by carefully restricting supply henceforth, may earn monopoly rents. In this manner, an otherwise renewable resource is converted to a non-renewable, storable resource. This set-up is akin to a cartel-and-fringe design, with poachers as the competitive fringe. By promoting the depletion of wild populations in the short run, the speculator eradicates potential competition from the fringe and so becomes a monopolist in the long run.[3]

[1] The Yale Center for the Study of Globalization notes that a major force behind this alarming trend is consumer demand in Japan; indeed, Japan has threatened to ignore any listing of tuna under CITES. See <http://yaleglobal.yale.edu/content/adieu-atlantic-blue-fin-tuna> for details.

[2] This is related to, but different from, the physical concept of the minimum viable population (MVP) to indicate what safety margins should be respected to maintain 'acceptable' extinction probabilities for a certain time horizon.

[3] Bulte et al. (2003*b*) develop a similar idea, but their model was based on the unrealistic assumption of a speculator 'bribing' poachers to expand their supply. Mason et al. (2012) provide a formal technical analysis of incentive-driven extinction as a purposeful strategy by developing a more realistic model in which speculators purchase commodities on the black market. We provide an intuitive description that is based on the technical material in these earlier papers.

We analyse the potential profitability of banking on extinction. Two possible solution paths emerge from this discussion. Under the first, the speculator draws down his private stock before poachers start to harvest, operating as a traditional non-renewable resource monopolist constrained by potential entry. Under the second, the speculator actively participates in the market as a buyer, building up stores while at the same time encouraging poachers to harvest so rapidly as to drive the species below the MVP, thereby dooming it to extinction. Shortly thereafter the species becomes economically unattractive to harvest, and the speculator can operate as a monopolist with exclusive rights to a non-renewable resource. Which of these paths is more profitable depends on the initial levels of private and natural stocks.

The potential for a speculator to employ this banking on extinction strategy is particularly worrisome in cases where the extinct species is similar to a surviving species. Consider, for example, the black rhino. Legalizing trade in black rhino horns would reduce the transaction costs of trading this commodity, further increasing profits for the speculator. However, as horns of black rhinos and white rhinos are hard to distinguish, legalizing the black rhino horn trade would likely facilitate the laundering of white rhino horn and could therefore induce white rhino poaching. To avoid this unhappy outcome, a revision to the Convention that retains the trade ban for recently extinct but previously endangered species would seem prudent.

This chapter is organized as follows. Section 15.2 presents a brief review of the international regulation of trade in products derived from threatened and endangered species, as stipulated by CITES. Examples are provided to illustrate the economic scale of these activities. Consideration is given to the current research on the role of trade in the exploitation of endangered species. Section 15.3 describes a simple model of harvests from a renewable resource stock with two types of agents: poachers, who provide a flow of resource to consumers; and speculators, who are assumed to hold a pre-existing stockpile of the resource and can strategically manipulate market prices—and the threat of extinction—by accumulating or liquidating stores. A brief overview of the speculator's problem is given in section 15.4, followed in section 15.5 by an empirical application of the model to the trade in black rhino horn. Section 15.6 discusses policy implications of this model, as well as some of the deeper concerns surrounding the trade in products derived from endangered species. Section 15.7 concludes.

15.2 ENDANGERED SPECIES AND THE EFFECT OF TRADE

The extinction of key wildlife species and the decline of biodiversity in general remains a major concern for the global community. According to the

International Union for Conservation of Nature (IUCN) Red List of Threatened Species, one in three amphibians, one in four mammals, and one in eight birds are at risk of extinction in the wild (IUCN, 2012).[4] Using data for 25,780 vertebrates (i.e. mammals, birds, reptiles, amphibians, and fish), Hoffmann et al. (2010) estimate that one fifth of all species are classified as threatened (i.e. critically endangered, endangered, and vulnerable), and that, on average, fifty-two species of mammals, birds, and amphibians move one category closer to extinction each year. Most deteriorations in status class are reversible; however, 13 per cent of status class changes have resulted in species extinction. For example, at least two bird species and nine amphibian species became extinct between 1988 and 2008, and a further six critically endangered bird species and ninety-five critically endangered amphibian species became 'possibly extinct' during this period. No mammals are listed as becoming extinct during 1996–2008, although the one dolphin species (the Yangtze River dolphin, *Lipotes vexillifer*) has been flagged as becoming possibly extinct (Hoffmann et al., 2010).[5]

There are several proximate factors that contribute to the increased threat of extinction of wildlife species: the loss or degradation of natural habitats; the introduction of new species into a natural ecosystem; and the over-exploitation of a species for subsistence use, domestic commercial use, and international trade. Reid and Miller (1989) estimate that over-exploitation was the primary cause of extinction for 23 per cent of all mammal, 32 per cent of all reptile, 11 per cent of all bird, and 4 per cent of all fish species. The exploitation of wildlife for the international trade is much less important than subsistence use and domestic trade for the vast majority of wildlife species. However, for a few key species (e.g. primates, cats, elephants, rhinos, parrots, and reptiles), harvesting for international trade may be the primary factor threatening survival (Burgess, 1994).

CITES was established in the early 1970s. With 176 member states, it is the most important global initiative to monitor and regulate trade of nearly 35,000 species of animals and plants (CITES, 2012*a*). About 97 per cent of these species (included in Appendices II and III of CITES 2012*a*) can be traded under certain conditions set out by the Convention. The remaining 3 per cent of these species are generally prohibited from international commercial trade (Appendix I).[6] Globally, trade in key CITES-listed Appendix II animals and

[4] Walpole et al. (2009) describe the challenges and discuss the progress towards improving biodiversity indications and tracking targets at the global level.

[5] However, many sub-species of mammals have been declared extinct, including the eastern cougar (in 2011), the Japanese river otter (in 2012), the Pyrenean ibex (in 2000), the Sturdee's pipistrelle (in 2000), and the western black rhino (in 2011).

[6] CITES affords varying degrees of trade protection based on the biological status of wildlife species, and the extent to which they are threatened by trade. Member countries prohibit international trade in currently endangered species that are listed in Appendix I. Species that

their products is valued at US$350–530 million/year, and almost US$2.2 billion from 2006–10 (CITES 2012*a*).

There is also considerable illegal trade in Appendix I wild species and their products. It is difficult to obtain reliable figures for the volume and value of the illegal international wildlife trade, but they are thought to be substantial. For example, in September 2012 Malaysian custom officers confiscated approximately 700 African elephant tusks—worth about US$1 million—destined for China; this was the third substantial seizure of illegal ivory within three months (Wexler, 2011). Meanwhile, estimates of the price of ivory show a dramatic increase over three years, from approximately US$157/kg in 2008 to US$300–700/kg in 2012 (Wexler, 2011). Similarly, the retail price of tiger skin has risen to US$20,000/skin, and rhino horn to at least US$60,000/kg (Choi, 2010). Poaching for the illegal trade has been identified as the main cause of the 97 per cent decline in the population of African black rhinos since 1960 (IUCN, 2011*a*). Illicit wildlife trafficking is now recognized as a new form of transnational organized crime (TRAFFIC, 2012). At a November 2012 CITES event on 'Wildlife Trafficking and Conservation: A Call for Action', the US Secretary of State Hillary Clinton brought considerable attention to the need to combat the illegal trade in wildlife (CITES, 2012*b*).

The trade in wildlife and its commodities does not, in itself, necessarily lead to the decline in wildlife populations. For instance, Barbier et al. (1990) employ a bioeconomic model showing the conditions under which there exists an incentive to harvest wild species to the point of extinction. If wildlife species and their trophies are considered to be a valuable asset, then it may be in the interests of those responsible for the wildlife stock to maintain the renewable wildlife resources as they increase in value over time.

The relationships between the price of the traded wildlife resource, the costs of harvesting, and the return on comparable investments (i.e. the prevailing interest rate) determine the rate at which individuals decide to use the resource. However, from the standpoint of the individual exploiter, it may be optimal to harvest a species to extinction if there is a combination of:

(i) A high price of the resource relative to the cost of harvesting; and
(ii) A high net effective discount rate (i.e. actual discount rate adjusted for any real price increase) by users relative to the species growth rate.

may become endangered in the near future are listed under Appendix II. For these species trade is limited and monitored closely: export quotas depend on the biological status of the Appendix II species population in each producer country. Appendix III listing allows countries to prohibit trade in nationally protected species. However, member countries are permitted to take out 'reservations' against the Appendix listing, which allows them to continue trading in any species, even those threatened with extinction. Phelps et al. (2010) note that CITES needs to be strengthened, with improved data collection, analysis, and review, to ensure biodiversity protection.

As the costs of harvesting plants and animals from the wild are often extremely low in comparison with the price of the traded species or product, lucrative profits can be derived from exploiting the resource. While high economic rents alone do not necessarily create an incentive to over-harvest an endangered species, if these are combined with a situation where the net effective discount rate exceeds the growth rate of the harvested population, then it may be in the interest of the individual to deplete the renewable resource as quickly as possible, even to extinction (Barbier et al., 1990). Many wildlife species threatened by exploitation for the international trade—such as primates, rhinos, elephants, parrots, western Atlantic blue fin tuna, and larger cats and other fur-bearing carnivores—are slow to mature or have low reproduction rates (Oldfield, 1989); thus, this risk is tangible.

Species harvesting is considered to be an economic problem if the level of wildlife exploitation exceeds the 'socially efficient' level. There are several market and policy failures that may distort the incentives for conserving wildlife, drive a wedge between private and socially optimum levels of species exploitation, and lead to excessive species exploitation. For example, market failures exist when market prices fail to fully reflect environmental values. The presence of weak property rights, public environmental goods, environmental externalities, incomplete information and markets, and imperfect competition all contribute to market failure. Policy failure occurs when the public policies required for correcting market failures over- or under-correct for the problem. They also occur when government decisions or policies—in areas where there are no market failures—are themselves responsible for excessive exploitation of endangered species. The result of market and policy failures is a distortion of economic incentives; that is, the private costs of exploiting wildlife do not reflect the full social costs.

The underlying causes of the increased vulnerability of traded wildlife species are a mixture of market and policy failures that create incentives for excessive wildlife harvesting. In this chapter, an additional aspect of the market and its impact on incentives for excessive wildlife exploitation will be examined: that is, market power and speculation, where there exists an incentive to deplete a renewable wildlife resource to the point of extinction, in order to maximize profits from wildlife and their products over time.

15.3 A SIMPLE MODEL[7]

Our model includes two types of economic agents. One agent, whom we refer to as the speculator, holds a pre-existing stockpile of the resource. Other

[7] This section provides a heuristic description of the technical model in Mason et al. (2012); interested readers may find a more detailed discussion of the analytics there.

agents are poachers. Poachers harvest the resource under conditions of open access, so that instantaneous profits are always competed away. One important distinction between our model and the traditional open-access model is that the speculator can induce poachers to harvest more rapidly by adding his demand to market demand. The motivation for undertaking such behaviour is the possibility that it will lead to sufficiently rapid harvesting so as to doom the resource to extinction. Following extinction, the speculator acts as a monopolist, extracting from his stockpile in a fashion analogous to an exhaustible resource monopolist.

We assume that wild animals and supply by speculators are perfect substitutes.[8] Aggregate deliveries to market, therefore, are the sum of aggregate poacher harvests plus net deliveries from the speculator's stockpile. The change in the speculator's stockpile equals these net deliveries: if he sells, the stockpile shrinks; if he buys, the stockpile grows.

Poachers' revenues are determined by inverse market demand (i.e. marginal willingness to pay) by private individuals aside from the speculator. We regard this inverse demand as net of any anticipated penalties that might be imposed upon private individuals—for example, because of the possibility of confiscation, fines, or other sanctions. Unit harvest costs depend on the natural stock, with larger stocks lowering costs.[9]

[8] The speculator's supply may come from stockpiles of a storable commodity (such as ivory or rhino horn) or from captive animals (bears, rhinos). In reality, wild animals and the speculator's supply may be imperfect substitutes.

[9] The unit cost is derived based on individual poachers' costs along with profit-maximizing and market-clearing conditions. Specifically, an individual poacher's cost of harvesting, $c(x,S)$, is a declining function of the natural stock of the resource, S, and an increasing function of harvesting level, x. The marginal cost of harvest is positive, and may be constant or increasing. Individual poachers' harvests are profit-maximizing, so that marginal cost is equated to average revenue. If costs are linear in harvest, so that marginal cost is constant, then the individual poacher's optimal harvest is not determined (though aggregate harvest would be). If marginal costs are increasing, then the individual poacher's optimal action is well-defined for any combination of price and stock. In turn, this relation induces a supply curve for poachers, which determines aggregate harvest. Because of the open-access condition, aggregate harvesting levels adjust at each instant so as to make the typical poacher's costs equal to its revenues, which implies that price equals average cost. Between these two observations, we infer that equilibrium harvests lead to a condition where each poacher operates where marginal cost equals average cost (which equals minimum efficient scale in the event that marginal costs are not constant). Whether marginal costs are constant or increasing in harvest, the level of average cost that equals marginal cost is uniquely determined by stock size; this common level of marginal and average cost is $c_a(S)$. Given our earlier assumption that an increase in natural stock leads to lower costs for a given level of harvest, an increase in natural stock lowers unit cost at minimum efficient scale (i.e. $c_a'(S) < 0$). Finally, one might think of costs as implicitly including potential penalties associated with detection. With this interpretation, expected penalties would be akin to a tax on producers. As is well known, the same market outcome obtains whether a tax is imposed on buyers or on sellers. In our application, it is more convenient to model the 'tax' as being paid by buyers.

The number of poachers adjusts to set poacher economic profits to zero (i.e. price equals average harvest cost). The equilibrium condition for poachers therefore implicitly determines equilibrium net deliveries as a function of the natural stock; relatedly, equilibrium aggregate harvest is a function of the natural stock and net speculator sales. It follows that each one-unit increase in speculator sales is offset by a one-unit reduction in poacher harvests, leaving price unaltered.

The natural stock of the resource adjusts over time in the usual fashion, with the rate of change equal to gross additions to biomass less total harvest. Gross additions depend on the current stock of the resource, as described by natural recruitment. We assume there is a critical mass or MVP, such that an *in situ* stock shrinks inexorably if it ever falls below the MVP. There is also a larger value of stock, which can be interpreted as the carrying capacity of the resource, at which gross additions fall to zero. For levels of the resource between the critical mass and the carrying capacity, recruitment is a strictly positive, concave function of stock. One of the main points we will develop is the possibility that the speculator may strictly prefer a time path of purchases that forces the natural stock below MVP, even though stock would not fall so low in the absence of such behaviour.[10]

While the assumption of myopic behaviour associated with open-access harvests is analytically convenient, it is possible that in reality a cohort of forward-looking poachers stores some of their harvest, in an attempt to capitalize on future extinction.[11] To facilitate our discussion, we assume that there are sufficient barriers to entry into speculative markets as to insulate the speculator from future competition. Such barriers might be formed by set-up costs or asymmetric information, entry deterrence by the incumbent (Mason and Polasky, 1994), but also by moral or ethical considerations—the illegality of the trade suggests that most people will resist entering this business even if it implies forgoing monetary gains, akin to limited entry in drugs trading. Alternatively, the pre-existing stock of commodities owned by the speculator may be a decisive factor. An extinction strategy would increase the value of this extant stockpile, potentially making the extinction strategy a profitable undertaking for the speculator (but not for poachers with zero initial stocks). The assumption of a monopolistic speculator implies that we offer a discussion of the polar extreme case from Kremer and Morcom (2000), who assume instantaneous entry and exit in response to profit differentials, and model all agents as atomistic. However, it is important to realize that the key element

[10] Implicitly, we are assuming that conservation efforts (enforcement, investments in population or habitat, etc.) do not intensify as the stock approaches MVP. This seems like a natural policy response to impending extinction. We discuss this point in the concluding section.

[11] The solution to the dynamic problem of the speculator does not rule out upward price jumps; hence there appears to be scope for agents to arbitrage gains by 'entering the market post-extinction'. Gaudet et al. (2002) and Karp and Newbery (1993) consider similar scenarios.

driving our result is not the literal monopoly assumption, but the much less restrictive assumption of market power. The downward-sloping demand function the speculator faces can equally well be thought of as residual demand in the context of a cartel-and-fringe model. Indeed, the main message of this chapter could be reinforced in such a setting.

The speculator's flow pay-offs from transactions are positive if the speculator sells, and negative if he purchases. His goal is to maximize the present value of net benefits over time by choice of sales and purchase rates.

15.4 SOLVING THE SPECULATOR'S PROBLEM

There are two possible outcomes to consider. First, the speculator may pursue a *banking on extinction strategy*, in which he first adds to his stockpile (driving up prices, encouraging extra poaching, which helps drive the resource to extinction), followed by a phase in which he sells his stockpile as a monopolist. Especially if the speculator has access to a 'large' initial stock of the wildlife commodity, it may pay to hunt the wild stock to a level below MVP so that extinction becomes inevitable. As already mentioned, in addition to the benefits from unfettered market power, the speculator may enjoy an additional bonus from following the banking strategy. Insofar as current (international) trade in the species' commodities is banned by CITES, the trade ban might be lifted after extinction (as CITES regulates trade only in endangered species—not extinct ones: see Bulte et al., 2003*a*). Relaxation of a trade ban would lead to increased demand, raising profits from banking.

Second, the speculator may forgo the banking option and, instead, follow a *dumping strategy*. This implies divesting the stockpile while competing with poachers—a classical cartel-and-fringe set-up. The speculator will not drive poachers out of business, unless he decides to dump his stockpile on the market at once, which would be the optimal thing to do if price rises too slowly to make holding the stockpile a worthwhile investment. Keeping poachers out of business over a longer interval is not an optimal long-run strategy, as this involves selling off larger quantities of the stockpile at a lower price. Furthermore, the dumping strategy may or may not coincide with extinction of the wild stock; if extinction does occur, this outcome is due to poaching pressures (open-access harvesting) and has nothing to do with the speculator.

The choice of whether to pursue the banking strategy is a numerical one, and requires comparison of the present value of net benefits to the speculator under the banking and dumping strategies. In both cases, value is determined by the initial level of wild stocks and the initial level of private stores. The main point of this chapter is that, for appropriate combinations of these two initial levels, it pays the speculator to pursue a banking strategy, taking actions that lead to extinction of the wild stock. This choice does not require the initial natural stock to be lower than the MVP; however, banking is more attractive as wild stocks draw nearer to the MVP, as this would lower extinction costs. We now turn to a numerical example to examine whether banking on extinction may be a real threat for some species.

15.5 EMPIRICAL ILLUSTRATION: BANKING ON BLACK RHINO EXTINCTION

We now explore the profitability of banking on extinction by analysing whether the gain in the speculator's profits due to extinction is sufficient to cover the purchase costs. The specific example considered is that of the black rhino, for which data are available and there is evidence that speculation does exist. Our use of this case is not meant to imply that the banking outcome is inevitable for this species, but rather to demonstrate that it is optimal under a certain set of empirically defensible parameters. In this application we use data provided by Milner-Gulland and Leader-Williams (1992) and Brown and Layton (1998, 2001). Assuming that stockpilers care about conservation of rhinos, and are willing to forgo some profits to achieve that objective, Brown and Layton demonstrated that *ex situ* stocks of rhino horn may be used to promote rhino conservation (see also Fernandez and Swanson, 1996).[12] Here we demonstrate the opposite result: a profit-maximizing speculator who holds a sufficiently large private stock may trigger rhino extinction.

Private parties, mainly in Asian countries, have stored large quantities of rhino horn over the past few decades. Presumably, these stocks are held in the expectation that prices will rise rapidly enough to compensate for the interest income forgone (Hotelling, 1931). In the recent past, speculators have been proven right; rhino horn prices have increased tremendously since the mid-1970s and, according to one estimate, rhino horn now fetches up to US$60,000/kg in Asian markets (Choi, 2010). Such rapid price increases are more than enough to compensate for the lost interest that would accrue to immediate sales, justifying stockpiling of rhino horns. Since the 1970s, the wild population of black rhinos has collapsed from 65,000–100,000 animals to just about 2,500 in the 1990s, after which they stabilized at a level of about 4,000–5,000 rhinos at present. Unfortunately, poaching pressure has intensified in recent years, and

[12] Specifically, Brown and Layton demonstrate that by supplying from stores, rhino horn prices will fall such that poachers will exit. In the meantime, conservation efforts should be geared towards ensuring a sustainable supply of horn from 'cropping' rhinos bred in captivity to ensure that prices stay sufficiently low in order to dissuade renewed entry when stocks run out. Private speculators then have no choice but to liquidate their stocks, further depressing prices.

Table 15.1. Numerical analysis of banking on extinction for the case of rhino poaching

Discount	PVNB from banking	PVNB from dumping	Net gain from banking
Rate (%)	(US$ million)	(US$ million)	(US$ million)
5	50.03	26.43	23.60
10	34.11	22.23	11.88
40	10.9	9.8	1.1

well-equipped, sophisticated crime syndicates have killed more than 800 African rhinos between 2008 and 2011 (IUCN, 2011*a*). Although legal trade in rhino horn has been banned since 1977, a lucrative and well-established underground trade still exists and is the leading cause of the species' demise.

Currently, private stockpilers hold larger quantities of black rhino horn *ex situ* than wild stocks carry *in situ*. Speculators hold approximately 20,000 kilograms of rhino horn (Brown and Layton, 1998, 2001); we assume that these stocks are held by one agent.[13]

Based on this parameterization, we conducted a simulation analysis. Results from these simulations are presented in Table 15.1. Here we tabulate the net present value (PVNB) of the banking strategy (second column), and the dumping equilibrium (third column), and the net gains from the former over the latter (fourth column).

The PVNB of the banking scheme represents the discounted flow of monopoly profits, less purchase costs. While these costs can in principle be considerable, the first column indicates that they are more than offset by the post-extinction profit flow. This appears to hold for reasonable discount rates; in particular, it is true in our simulations at rates below 40 per cent.

The dumping PVNB summarizes similar statistics for the case where speculators face competition from poachers harvesting the wild stock. When speculators supply from private stores, they depress prices and temporarily push poachers out of the market; this allows rhino populations to recover, thereby reducing the unit cost of harvest. Ultimately, poachers return to the market; in anticipation of this fact, the speculator exhausts his stocks at the moment entry is triggered.

[13] Equivalently, one could imagine multiple speculators that collude as a monopolist in the pursuit of the banking on extinction strategy. Even if these agents interacted non-collusively, results similar to those we investigate could emerge. Such a scenario is more complicated to analyse, in that each agent ought to take other agents' strategies into account; we would then have to solve for the equilibrium to a differential game. While such a scenario is undoubtedly more realistic than our model of monopoly behaviour, the fundamental economic ingredients remain: when speculators have some ability to influence market price and can induce more rapid harvesting by poachers by offering bribes, it can pay them to drive the natural stock to extinction.

In the column labelled 'Net gain from banking', we deduct the dumping profits (column 2) from the banking profits (column 1) to obtain an estimate of the net profits of the banking strategy. The results suggest that gaining a (temporary) monopoly is profitable for a wide range of discount rates. Accordingly, we conclude that banking on extinction can represent a profitable strategy, if the private stockholder is not too impatient. Explicitly incorporating stores and speculators thus reverses the insights of traditional renewable resource models, and suggests that the rhino population is far from safe.

This brings us to an interesting and perhaps counterintuitive result. In our model, the extinction probability of the rhino is an *increasing* function of its intrinsic growth rate, which is opposite to the predictions of standard renewable resource models without storage and speculation (e.g. Clark, 1990; Swanson, 1994). The reason is that a high intrinsic growth rate lowers the profitability of dumping. It advances the date at which re-entry by poachers occurs, which requires more rapid depletion of the private stock (lowering prices in every period where the speculator is selling, and so reducing the PVNB from the dumping strategy). As a robustness check we have computed what happens when the intrinsic growth rate is doubled. This reduces the PVNB from dumping from US$26.4 to US$19.9 million, and leaves the PVNB of the banking strategy unaffected (because banking requires an initial 'cull' of the herd below the MVP, rendering the natural regeneration rate irrelevant). As a result, the net gain from the banking strategy, relative to dumping, increases to more than US$30 million. For speculators adopting a dumping strategy, living and growing rhino populations are a nuisance.

A similar story holds with respect to the discount rate. Conventional wisdom implies that high discount rates discourage investments in wild stocks and thus promote extinction, at least when populations are optimally managed (Clark, 1990). Not so when we account for the incentives of speculators. We find that the relative appeal of extinction decreases as the discount rate increases. Under the dumping strategy the benefits are realized in earlier periods, and favoured with high discount rates. In contrast, under banking on extinction the costs are immediate and the benefits are realized in the future. In other words, 'banking' compares favourably to 'dumping' when discount rates are low.

Our analysis considered black rhino conservation and exploitation in isolation. In reality, another species produces a close substitute for black rhino horn—the more common and docile white rhino. If one included white rhinos in the analysis, would banking on extinction still pay off? While this extension is beyond the scope of this chapter, we can imagine some of its features. Including white rhinos would raise the natural stock, and so would increase a speculator's costs of banking on extinction. On the other hand, this large extra harvest would raise the speculator's stockpile, with attendant benefits. There are additional, subtle, changes as well: because white rhino horns are not perfect substitutes for black rhino horns, one would need detailed information about cross-elasticities.

Furthermore, because white rhinos are more docile than black rhinos, it seems unlikely that the harvest costs to poachers are identical for the two species. A careful analysis would require information on all these points.

15.6 POLICY LESSONS

A number of recommendations follow from this analysis. First and foremost, the risk of a banking–extinction strategy can be attenuated if wild stocks remain sufficiently large. That is, if anti-poaching conservation efforts manage to steer wild populations away from MVP levels, then the costs of pursuing a banking strategy increase. Therefore, it is in the interests of host countries and the international community to increase their investment in conservation efforts of endangered species to ensure that populations remain sufficiently large and robust. Underscoring the timeliness of our story, the Security Council of the United Nations recently called for an investigation into elephant poaching.[14]

Second, the relative appeal of the banking strategy will be diminished by actions that reduce its present discounted value. The necessary reduction is not to zero, but to the level associated with the dumping strategy. This reduction can be realized, for example, by lowering flow profits—either by efforts aimed at lowering demand (i.e. moral suasion) or by lowering the quality of the product.[15] But the requisite reduction in value can also be obtained by increased enforcement, either raising poachers' costs, or by removing the feature of current regulations that drops trade sanctions if the species becomes extinct. This latter action, which is relatively low-cost, seems like an obvious place to start.

Additionally, CITES needs to pay much greater attention to the economic incentives created by its trade intervention policies. Banning trade while keeping demand unchecked, for example, encourages stockpiling of wildlife commodities. Our analysis suggests that, as stocks of such commodities grow over time, they could evolve into a liability for conservation: large *ex situ* stocks increase the profitability of an extinction strategy. Moreover, because trade in *extinct* species is legal, owners of large stockpiles may find it

[14] See <http://cites.org/eng/news/pr/2012/20121222_UNSC_elephant_LRA.php>.

[15] One way to obtain this outcome is to take actions that impact the ability of the animal part to achieve its ultimate goal. If the use is medicinal in nature, one could imagine capturing the animals, sedating them, and then sprinkling some agent on the animal part. Obviously, the introduced agent would have to be benign from the animals' perspective. If the ultimate goal has to do with appearance, then disfiguring the part—for example, by spray-painting it, would do the trick. This sort of action has been used to diminish the value of certain seals, thereby lowering the value of their fur coat.

worthwhile to promote an extinction strategy so as to remove the trade ban.[16] If so, CITES has inadvertently created the context in which extinction is promoted, rather than prevented. CITES may need to consider removing this loophole, for example by creating an Appendix 0 to ban trade in certain extinct species (Bulte et al., 2003*a*).

CITES or host countries may also need to consider market intervention to reduce the surging prices (albeit through illegal trade) for key endangered wildlife commodities. For example, it seems prudent to regularly convert such private stocks into a conservation asset for the international community, via public purchase programmes or controlled auctions, for example (Kremer and Morcom, 2000; Bulte et al., 2007). This would both raise funds for conservation and lower the market price for the wildlife species and its parts, thus reducing the incentive for harvesting the wild species. However, such sales of stock are not currently an option given the Appendix listing and trade restrictions adopted by CITES.

In addition, to attenuate incentives to bank on extinction, the international community might invest in securing alternative sources of supply of wildlife products. If substitute products (farmed, synthetic or otherwise) are made available, potential monopoly rents are curtailed. One potential alternative source of wildlife products could be a flow of commodities from farmed wildlife species. For example, in China bears are farmed for their bile, and there are several officially sanctioned tiger farms for skins and bones (Chunyu, 2011). Similarly, in Texas, a number of species that are currently extinct in the wild (e.g. scimitar oryx) are bred on privately held ranches for the purposes of providing game-hunting opportunities. While controversial, these activities have led to significant population expansions (CBS News, 2012).[17] However, such approaches raise other ethical questions about wildlife conservation and care, and it is not clear whether captive alternatives complement or undermine the conservation of extant wild populations.

Next, insofar as the objective is to remove the incentive to bank on extinction, it would be particularly worthwhile to focus on lowering the backstop price of wildlife commodities—for example, by using information campaigns to reduce demand. Such a campaign would be particularly efficient if it targeted those consumers with the highest marginal utility of consumption, shifting in the demand curve at higher prices. However, demand reduction will take time, because cultural attitudes and historical beliefs are difficult

[16] Bulte et al. (2003*a*) demonstrate that under some conditions the value of stockpiled ivory in Africa is sufficiently high to make a strategy aimed at driving the African elephant to extinction financially viable.

[17] However, these activities have recently been ruled to conflict with the 1973 Endangered Species Act, and many ranchers have indicated that they plan on abandoning breeding and conservation efforts as a result (<www.mysanantonio.com/news/local_news/article/Hunting-ban-could-see-last-of-unicorns-3453819.php>).

to change. In addition, the final demand for many wildlife products is rooted in the Asian market, which is rapidly expanding due to a combination of population and income growth.

A second information-related policy option may be worth exploring. The 'banking on extinction' strategy depends on consumers treating stockpiled wildlife products—or those recovered from captive animals—as reasonable substitutes for poached products. If banking strategies can be identified as primary drivers of poaching activities (rather than immediate supply strategies), it may be possible to highlight knowledge of chemical breakdowns or other quality differences between 'fresh' and stockpiled sources (or, in the case of bear bile, for instance, products derived from captive vs. wild bears). If made aware of differences, consumers may favour wild sources. In theory, this could reverse the stockpiling incentives of major, forward-looking speculators to favour wild conservation.

Extending this line of reasoning is the possibility of undermining the entire market for illegally traded wildlife by creating a large-scale 'lemons' problem. Illegal trade in wildlife is already wrought with informational problems (Beckert and Wehinger, 2012). Intermediaries and consumers must often rely on perceived trustworthiness of suppliers when purchasing wildlife products, particularly those which come in a final form that is difficult to identify or are associated with no known biological response in humans (e.g. powdered rhino horn). One could imagine a collaborative effort by researchers and governments to develop inexpensive processes for synthesizing imitation rhino horns that are indistinguishable from the real thing, for the purposes of exacerbating informational problems. Already, some rhino poachers have been detained with elaborately produced fakes that are visually identical, but easily distinguished through basic chemical tests (e.g. burning flakes), and various imitation production facilities using bovine bones have been established on the outskirts of Hanoi to fuel Vietnamese demand (Milliken and Shaw, 2012). If it is possible to build on these efforts and produce specimens that are very difficult to distinguish from the real thing at all retail levels, informational market failures may worsen, undermining returns to poaching. Rather than marketing these substitutes to consumers (as already noted, consumers may be uninterested in alternatives, regardless of how similar they are to the real thing), the formulas could be released to the public domain for enterprising individuals to take advantage of. As fakes flood the market prices will fall, and identifying legitimate specimens will become more difficult.[18]

Finally, a major overhaul of the existing approach to wildlife conservation and the trade in endangered species and their parts may be required. In 'Elephants, Economics and Ivory', Barbier et al. (1990) ask the question: 'Are

[18] This strategy has risks, as it has the potential to embolden organized crime or broaden the consumer base for poached products; it may also perpetuate the perception of medicinal benefits.

elephants worth more dead than alive?' This question continues to be relevant to the current wildlife debate, where numerous endangered wildlife species are in high demand for their commodities, there are often ambiguous property rights for the wildlife species, negative incentives for conservation may exist due to human–animal conflict, and inadequate enforcement prevails. Legal, regulated trade needs to be carefully considered as an option to reduce the poaching incentives, increase investment in enforcement, and provide compensation for wildlife damage. It is not infeasible that, in the longer term, the highly valuable natural wildlife stocks may be able to contribute a steady flow of funds for investment in natural and human capital, and enable host countries to achieve broader sustainable economic development objectives.

15.7 CONCLUSIONS

Wildlife commodities harvested in nature and those sold from either private stockpiles or farms (captive breeding) compete on output markets. When these private stockpiles are sufficiently important, they can create an incentive to promote extinction of wild stocks, after which the speculator earns monopoly rents. Our simulation study of rhino horn storage indicates that current *ex situ* stockpiles are sufficiently large that profit-maximizing individuals may have an incentive to subsidize the slaughter of rhinos until the wild stock collapses.

'Banking on extinction' might pose a real threat to conservation of certain rare species providing valuable and storable commodities. Of course it is an open question to what extent the numerical results of the rhino case could apply to other species. We speculate that for some species they might. For example, bear bile prices have increased to incredible levels in response to increasing scarcity of bear gall bladders—Mills, Chan, and Ishihara (1995) state that prices paid in South Korea went up to $210,000/kg. Chinese investors keep nearly 10,000 bears on so-called bile farms, where bile is drained from live bears through devices surgically implanted in their gall bladders. It may be profitable for these investors to promote extinction of wild stocks as this would increase their market power and moreover relax existing international trade restrictions (most of the world's bear species are listed in Appendix I of CITES). Bear (or tiger) farming implies that speculators 'own' a renewable resource, rather than an exhaustible stockpile of a commodity such as rhino horn or ivory. This implies that they are able to enjoy monopoly rents for a longer, indeed potentially infinite, period, which enhances the profitability of banking on extinction.

Ultimately, whether any other species is likely to be the victim of a speculative attack is an empirical matter. Our point is that such a gloomy scenario should not be regarded as empirically irrelevant. Moreover, some of the policy implications of our model run counter to some existing insights. While Kremer and Morcom (2000) and Brown and Layton (1998, 2001) consider *ex situ* stockpiles of wildlife commodities to be assets that could be strategically used to enhance conservation, we point out that they are potentially dangerous liabilities when in the hands of profit-maximizing individuals. Therefore, from a conservationist perspective it makes sense to promote the transfer from such stocks from private to public parties—either through confiscation or purchase.

Finally, in an interesting twist to this analysis, we would like to note that there are conceivable cases where the interests of conservationists and speculators run parallel. Speculators care only about restricting supplies from the wild, and presumably are equally happy with a well-enforced harvest (or trade) ban as with extinction. When public agencies can commit to strict conservation, the incentive to bank on extinction evaporates.

APPENDIX: CALIBRATING THE NUMERICAL MODEL

We interpret the recent stabilization of rhino abundance as a sign that the dynamic system has reached a new steady state, in which poachers earn zero profits and where replenishment of the rhino population exactly equals harvesting. Assuming that open-access harvesting has reduced the rhino population to such a bio-economic equilibrium, with 2,500 rhinos, one can solve for equilibrium growth and harvests, equilibrium effort levels, and costs per unit of poaching effort. Storage costs are negligible when compared with the value of rhino horn, and are hence ignored in what follows. Throughout we assume that one rhino carries 3kg of horn.

We first define a (skewed) logistic growth function $F(S) = 0.16S[1-(S/100{,}000)^7]$, where S is measured in rhinos (see Brown and Layton (2001), hereafter B–L). Since we are interested in studying extinction and near-extinction of rhinos, we explicitly introduce the minimum viable population (MVP) concept. We 'shift down' (or horizontally to the 'right') the growth function as just defined by a constant so that it intersects the horizontal axis ($F(S) = 0$) at stock levels somewhat greater than zero (and somewhat smaller than K). We assume that 100 rhinos is a reasonable estimate for the MVP (Primack, 1998).

In equilibrium, 352 rhinos are harvested, with a long-run stock of 2,500. Based on results in Milner-Gulland and Leader-Williams (1992) (hereafter MG–LW), equilibrium poaching effort can be calculated as 542 units. To determine the per-unit cost of poaching effort, we need to know the demand for rhino horn. Data on supply and rhino horn prices are difficult to obtain since the trade moved underground in the late 1970s. While very little information exists about the 'backstop price' of rhino horn (i.e. the price where demand is reduced to zero), some data are available for 'intermediate' output levels. Specifically, according to B–L, 8,000 kilograms were traded at \$168/kg and 3,000 kilograms were traded at \$1,351/kg. Using these observations, we assume a log-linear inverse demand curve $P(Q) = be^{-aQ}$, where b = \$4,719 is the backstop price, and a = 0.00042 is a parameter measuring the curvature and slope of the demand curve. With this specification, price is \$2,945 at harvest levels of 352. Following B–L, we assume that poachers receive only 3/8 of this price, so that the cost of organizing a poaching trip assuming zero profits is \$709. This number is somewhat larger than cost estimates provided for rhino hunting in Zambia by MG–LW, but may be interpreted as an aggregate cost, combining both the 'true effort' cost and an expected fine or penalty (treated separately by MG–LW).

Finally, the numbers in this chapter are based on the parameterized model by MG–LW. B–L, in contrast, assume that kills per expedition can be approximated by a constant, which will result in somewhat biased outcomes if the rhino stock changes over time and is an input in production.

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