1. Introduction

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ECOSYSTEM SERVICES: METHODOLOGICAL ISSUES, VALUATION AND POLICY

Biodiversity and ecosystems are degrading fast. Although the Sixth Conference of the Parties (COP6) to the Convention on Biological Diversity (CBD) held at The Hague, Netherlands in 2002 set a target of achieving a significant reduction in the rates of biodiversity loss by the year 2010, evidence shows that these rates have remained steady, if not accelerated. For instance, the Millennium Ecosystem Assessment (MEA, 2005) suggests that future extinction rates will rise to more than 10 times higher than current extinction rates that are 1000 times higher than fossil records of less than one species per 1000 mammal species becoming extinct every millennium (Table 1.1). The IUCN's Red List suggests that about 14 per cent of bird species, 22 per cent of mammals and 31 per cent of amphibians are threatened with extinction over the next century. The Living Planet Index – a measure of the state of the world's biodiversity based on trends from 1970 to 2008 covering 1432 terrestrial species, 675 marine species and 734 freshwater species – indicates an overall decline of 30 per cent in the global living planet index over this period (Table 1.1). This decline was steeper (60 per cent) for tropical species. The Ecological Footprint – a measure of humanity's demand on the earth's biocapacity for meeting consumption needs and absorbing wastes – has exceeded the earth's biocapacity by more than 50 per cent as of 2008.

The MEA (2005) observes that between 30 to 50 per cent of different ecosystems such as mangroves, coral reefs, wetlands and global forests have disappeared or degraded (Table 1.2). Further, they note that about 60 per cent of the world's ecosystem services are degraded. Of 24 ecosystem services reviewed only four services, namely, crop, livestock and aquaculture production, and carbon sequestration have improved, whereas the remaining 20 services have degraded or declined. The MEA (2005) emphasizes the importance of biodiversity to ecosystem services, human well-being and sustainable development.

Table 1.1 Evidence regarding rates of biodiversity loss estimated by different agencies

Indicator/Source	Explanation	Biodiversity loss
MEA, 2005	Current Extinction Rates	1000 times higher than fossil records of: <one species per 1 000 mammal species becoming extinct every millennium
	Projected Future Extinction Rate	>10 times higher than current rates
IUCN's 2008 Red List	Since 1500 AD:	
	Documented Extinctions	804
	Extinctions in the wild	69
	Species threatened with extinction over next century (IUCN's 2008 Red List based on an assessment of <2.7% of the world's 1.8 million described species)	16 928 14% of bird species 22% of mammals 31% of amphibians
Living Planet Index (WWF and ZSL, 2012)	A measure of the state of the world's biodiversity based on trends from 1970 to 2008 covering 1432 terrestrial species, 675 marine species and 734 freshwater species	Overall decline of 30% in the global living planet index over the 38-year period under review and similarly for terrestrial, marine and freshwater species.
Ecological Footprint	A measure of humanity's demands on the Earth's bio-capacity for meeting consumption needs and absorbing wastes	Humanity's Ecological Footprint has exceeded the earth's biocapacity by more than 50% as of 2008. In recent decades, the carbon footprint is a significant component (i.e., 55%) of this ecological overshoot.

Source: MEA (2005); for IUCN 2008 Red List, Vié et al. (2009); WWF/ZSL (2012)

Table 1.2 Evidence regarding rates of loss or degradation of ecosystems and ecosystem services

Evidence shows that:
35% of Mangroves
30% of Coral Reefs
50% of Wetlands
40% of Global Forests (in the last 300 years) have disappeared or degraded
About 60% of the world's ecosystem services are degraded
Of 24 ecosystem services reviewed, only 4 services i.e., crop, livestock, aquaculture production and carbon sequestration have improved:
Two other services, fisheries and freshwater, found to be beyond sustainable levels; remaining 18 services have degraded or declined

Source: MEA (2005)

The Tenth Meeting of the Conference of Parties (COP10) to the CBD held at Nagoya, Japan in 2010 formulated a strategic plan for biodiversity (2011–20) and 20 targets (the Aichi Biodiversity targets) which sought 'to initiate action to address the underlying causes of biodiversity loss including production and consumption patterns, by ensuring that biodiversity concerns are mainstreamed throughout government and society through communication, education and awareness, appropriate incentive measures and institutional changes'. The vision of the strategic plan is a 'world living in harmony with nature' where 'by the year 2050 biodiversity is valued, conserved, restored and wisely used, maintaining ecosystem services, sustaining a healthy planet, and delivering benefits essential for all people'. The plan envisions that by 2020 the rate of loss of all natural habitats are at least halved and, where feasible, brought close to zero, and degradation and fragmentation is significantly reduced. How to conserve biodiversity and ecosystem services therefore poses a challenge to planners and scientists.

Many ecosystem goods and services are not traded in conventional markets or are difficult to value. Hence, it is felt that if these values could be captured and internalized in decision-making it could lead to better conservation outcomes. Ecosystem services valuation (ESV) has, therefore, achieved considerable prominence in research and policy circles in recent years and especially after the publication of the MEA (2005). The Economics of Ecosystems and Biodiversity (TEEB) initiative by the United Nations Environment Programme (UNEP) and the EC (European Commission), and set up of the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) testify to the importance assigned to ecosystem services valuation. International conferences, special issues of journals such as *Ecological Economics* focused on ESV and the launch of a new journal *Ecosystem Services* bear this further.

In a seminal paper, Costanza et al. (1997) estimated the value of the world's ecosystem services at an average of over US$33 trillion (1994 US$). This is about US$49 trillion in 2010 US$ (Ninan and Inoue, 2013). Although their study received wide attention (over 11 200 citations as of April 2014), it was also widely criticized. For instance, it was noted that their estimates based on willingness-to-pay (WTP) measures were twice the global gross national product (GNP) of US$18 trillion per year and, further, that they ignored the ecological feedbacks and non-linearities that are central to the processes that link all species to each other and their respective habitats (Smith, 1997, cited in Ninan, 2011). Other problems such as double counting, ignoring substitution effects and budget constraints are also mentioned (Loomis et al., 2000). However, Costanza et al. have countered this by referring to the shortcomings of traditional GNP and WTP measures (Ninan, 2011). Notwithstanding these criticisms, their study led to an exponential growth in the literature in this area. The Ecosystem Valuation Toolkit (EVT), a bibliographic database of published ecosystem services papers lists about 44 000+ abstracts. However, this may also include papers that have not estimated the economic values of ecosystem services. The TEEB project (2008–10) database on monetary values of ecosystem services contains over 1350 data-points from over 300 case studies. Using this, de Groot et al. (2012) estimated the total mean annual value of the ecosystem services provided by different biomes in the world to range between 2007 international dollars (I$) 490 and 0.35 million per ha (Table 1.3). These values are the highest for coral reefs/coastal wetlands and lowest for open oceans. For tropical forests, these values are about I$5300/ha/year as compared to just over I$3000/ha/year for temperate forests.

The contribution of each ecosystem service to the total monetary value of ecosystem services shows wide variation across biomes (Table 1.4). For instance, for coral reefs, erosion prevention, recreation, provision of genetic resources, diversity and raw materials, and disturbance moderation account for the major share (over 97.5 per cent). For coastal wetlands, waste treatment and habitat services have the dominant share (92.5 per cent). These values for climate regulation are the highest for tropical forests (over I$2000/ha/year) and lowest for woodlands (I$7/ha/year). Intangible benefits and non-tradable public benefits account for the major share of the total value of ecosystem services across different biomes. The criticisms levelled against the estimates by Costanza et al. (1997) may equally apply to the estimates by de Groot et al. (2012). It is also not clear whether they have done a quality check of the reviewed studies. For instance, a recent

Table 1.3 Total monetary value of the bundle of ecosystem services per biome (2007 Int.$/ha/year)

	No. of estimates	Total of service mean values	Total of St. Dev. of means	Total of median values	Total of minimum values	Total of maximum values
Open oceans	14	491	762	135	85	1 664
Coral reefs	94	352 915	668 639	197 900	36 794	2 129 122
Coastal systems	28	28 917	5 045	26 760	26 167	42 063
Coastal wetlands	139	193 845	384 192	12 163	300	887 828
Inland wetlands	168	25 682	36 585	16 534	3 018	104 924
Rivers and lakes	15	4 267	2 771	3 938	1 446	7 757
Tropical forests	96	5 264	6 526	2 355	1 581	20 851
Temperate forests	58	3 013	5 437	1 127	278	16 406
Woodlands	21	1 588	317	1 522	1 373	2 188
Grasslands	32	2 871	3 860	2 698	24	5 930

Source: de Groot et al. (2012). Reproduced with permission from Elsevier

review cites several flaws of existing forest valuation studies such as being improperly designed and conducted, authors not even cross checking their calculations before publishing or mentioning the year to which their value estimates pertain (Ninan and Inoue, 2013).

Although economic valuation is a useful tool to aid decision-making and policy for conserving biodiversity and ecosystem services, we are aware of the limitations and criticisms levelled against economic valuation and conventional cost–benefit analysis (CBA), and the need for relying on plural approaches to justify conservation (Norgaard, 2010). Economic valuation, however, does not imply that other perspectives for better management of the environment have to be neglected. All that it seeks to convey is that if proper values are assigned to environmental goods and services it would lead to better conservation outcomes. However, market values tend to be precise monetary values but can be inaccurate as measures of social costs and benefits. Moreover, the estimated monetary values of ecosystem services are sensitive to changes in quantity, incomes, prices and methods used to value these services, time, space and contexts (Braat and de Groot, 2012).

Among the criticisms levelled against ESV is the lack of clarity about the definition of the term 'ecosystem service'. Costanza et al. (1997) defined it as 'the benefits human populations derive directly or indirectly from ecosystem functions', whereas the MEA (2005) defined it as 'the

Table 1.4 *Summary of monetary values for each service per biome (2007 Int.$/ha/year)*

	Marine	Coral reefs	Coastal systems	Coastal wetlands	Inland wetlands	Fresh water (rivers/lakes)	Tropical forests	Temperate forests	Woodlands	Grasslands
Provisioning services	102	55 724	2 396	2 998	1 659	1 914	1 828	671	253	1 305
1 Food	93	667	2 384	1 111	614	106	200	299	52	1 192
2 Water				1 217	408	1 808	27	191		60
3 Raw materials	8	21 528	12	358	425		84	181	170	53
4 Genetic resources		33 048		10			13			
5 Medicinal resources				301	99		1 504			1
6 Ornamental resources		472			114				32	
Regulating services	65	171 478	25 847	171 515	17 364	187	2 529	491	51	159
7 Air quality regulation							12			
8 Climate regulation	65	1 188	479	65	488		2 044	152	7	40
9 Disturbance moderation		16 991		5 351	2 986		66			
10 Regulation of water flows					5 606		342			
11 Waste treatment		85		162 125	3 015	187	6	7		75

12 Erosion prevention		153 214	25 368	3 929	2 607		15	5	13	44
13 Nutrient cycling				45	1 713		3	93		
14 Pollination							30		31	
15 Biological control					948		11	235		
Habitat services	5	16 210	375	17 138	2 455	0	39	862	1 277	1 214
16 Nursery service		0	194	10 648	1 287		16		1 273	
17 Genetic diversity	5	16 210	180	6 490	1 168		23	862	3	1 214
Cultural services	319	108 837	300	2 193	4 203	2 166	867	990	7	193
18 Aesthetic information		11 390			1 292					167
19 Recreation	319	96 302	256	2 193	2 211	2 166	867	989	7	26
20 Inspiration		0			700					
21 Spiritual experience			21							
22 Cognitive development		1 145	22					1		
Total economic value	491	352 249	28 917	193 845	25 682	4 267	5 264	3 013	1 588	2 871

Notes: Numbers in the cells are averages of the values found for a particular service and biome. Calculations based on a total of 665 values.

Source: de Groot et al (2012). Reproduced with permission from Elsevier benefits people obtain from ecosystems'. The MEA (2005) categorizes ecosystem services into provisioning, regulating, supporting and cultural services. However, some feel that apart from its other drawbacks, the MEA framework could result in double counting when a service is valued at two different stages of the same process providing human welfare (Boyd and Banzhaf, 2007; Ojea et al., 2012). Ojea et al. (2012) give the example of a forest providing water flow (regulating service) and water supply for hydropower (provisioning service). They also suggest that the MEA classification is not clearly focused on the final outcomes that ecosystem services provide to humans, which are what generate an impact (positive or negative) on human well-being and therefore have an economic value. Fisher et al. (2011) review definitions of ecosystem services and note that terms such as 'ecological functions', 'processes', 'services' or 'benefits' are employed with no clear definitions, and often refer to different concepts. To overcome this, some authors recommend distinguishing between benefits and services (Boyd and Banzhaf, 2007). Services are then considered as processes of ecosystems that relate to well-being, while benefits are outcomes of ecosystem services that have a direct relationship to human welfare. From the viewpoint of an accounting framework, some authors recommend a delineation of intermediate services, final services, and benefits as useful for valuation (Boyd and Banzhaf, 2007; Fisher et al., 2011). Boyd and Banzhaf (2007) illustrate this with a conventional market good such as a car. GDP only counts the car's value, not the value of steel used to make a car since this value is already part of the car's total value. They argue that the same principle holds for ecosystem services and give the example of clean drinking water consumed directly by a household, which is dependent on a range of intermediate ecological goods that should not be counted in an ecosystem service account. However, a lot of confusing terminologies are used which emphasizes the need for greater clarity about the definition of ecosystem services, especially from the viewpoint of valuation and human well-being.

Another criticism relates to the methods used to value ecosystem services. Although economic valuation tools are widely used to value non-market goods, they also have shortcomings. Direct market valuation approaches such as market-, cost- and production-based approaches use data from actual markets and hence are assumed to reflect actual preferences or costs to individuals. Market-price-based approaches are used to value tangible benefits or provisioning services of ecosystems, such as timber and non-timber forest products and fish resources. Cost-based approaches, such as the avoided-cost method, seek to answer how many dollars need to be spent to avoid the loss of an ecosystem service, or the replacement-cost method, which seeks to find out how many dollars have to be spent to replace the loss of an ecosystem service (e.g. the cost of chemical nutrients required to replace the loss of soil nutrients due to soil erosion). The mitigation cost method seeks to assess the cost of mitigating or restoring the loss or degradation of an ecosystem service. Cost-based approaches are often used to value regulating services such as soil and water conservation functions of forest watersheds. Production-function-based approaches use scientific information to establish a cause–effect relationship between the ecosystem service being valued and the output level of marketed commodities. Nahuelhual et al. (2007) used this approach to assess the relationship between drinkable water output and stream water flows and other parameters.

Revealed preference methods (RPM), such as the travel cost method (TCM) that is used to value recreational benefits and the hedonic pricing method (HPM) that is used to derive the implicit demand (or price) of an environmental attribute, are based on observations of individual choices and preferences as revealed in actual markets. TCM uses data on cost and the opportunity cost of time spent by an individual in visiting a recreation site to derive a consumer demand for visiting a recreation site. This is then used to analyse the value of a change in the provision of recreational benefits. HPM tries to utilize information about the implicit demand for an environmental attribute of a marketed commodity such as house or property prices. Property prices are influenced by several attributes including environmental attributes such as the proximity of a house to a forest, park or lake. A house or property located in the vicinity of a polluting factory or slum will command a lower price than one located near a park or lake. By estimating the demand function for a house or property, the analyst can infer the value of a change in the non-marketed environmental benefits generated by the environmental good (Pascual et al., 2011). An advantage of RPM is that it is based on individual choices and preferences as revealed in actual transactions. However, RPM methods estimate only use values, and are not designed to estimate non-use values. The estimated values may also be influenced by the technical assumptions regarding the relationship between the environmental good and the surrogate market good and hence sensitivity analysis to econometric functional form, value of travel time, etc. is often called for.

Stated preference methods (SPM) such as the contingent valuation method (CVM), choice modelling (CM) and group valuation (GV) simulate a market and demand for ecosystem services by means of surveys on hypothetical (policy-induced) changes in the provision of ecosystem services (Pascual et al., 2011). SPM are used to estimate both use and non-use values of ecosystems and/or when no surrogate market exists from which the values of ecosystems can be deduced. CVM uses surveys to ask people how much they are willing to pay to enhance the provision of an ecosystem service or alternatively how much compensation they would be willing to accept (WTA) for the loss or to avoid the loss or degradation of an ecosystem service. Unlike CVM, in CM individuals are asked to choose between two or more alternatives with shared and varying levels of attributes of the services to be valued. GV combines SPM with elements of deliberative processes. Though SPM have been widely used to value species, habitats and ecosystem services, and are the only known methods to value non-use values, they also have their limitations. Hypothetical bias, which occurs when the stated WTP differs from the actual WTP, has been found to be a frequent (but not universal) problem. However, several strategies have been developed since 1997 to minimize and in some cases eliminate this bias (Loomis, 2013). Embedding effect and insensitivity to scope are other problems in SPM. Despite these limitations, SPM are probably the most flexible methods for valuing many non-market ecosystem services.

Benefit transfer (BT) is a quick, inexpensive and practical method to estimate the value of ecosystem services of a policy site (a site where the values are to be estimated) by transferring values from a study site (a site where a primary valuation study has been conducted). This tool is particularly useful when, due to resource and time constraints, one is unable to conduct a primary valuation study in the proposed study area. In applying BT, however, due care has to be taken so that the policy and study sites closely match each other in terms of socio-economic and ecological contexts or values adjusted to reflect the important differences between the two sites. However, transfer errors and challenges in scaling up values, such as spatial heterogeneity, differences in contexts, availability of substitute and complementary sites and services, non-constancy of marginal values, distance decay, etc., can reduce the reliability of BT (Pascual et al., 2011).

Monetary valuation is problematic when critical thresholds are reached. Even a small disturbance can lead to a large change in the stock of an ecosystem service. In these cases non-linear valuation models are necessary to capture what may be exponentially rising values as critical thresholds are approached. There is also considerable uncertainty regarding the impact of dynamic factors, non-linearities in ecological processes, regime shifts and catastrophes on the flow and value of ecosystem services. Turner et al. (2003) emphasize that the explicit recognition of multiple, interdependent ecosystem services and values pose both conceptual and empirical research challenges.

Because of the limitations of economic valuation, some experts emphasize the need to rely on plural approaches to aid decision-making on conserving biodiversity and ecosystem services (Norgaard, 2010). In this context multi-criteria analysis (MCA) and deliberative processes have been suggested, in addition to economic valuation (Pascual et al., 2011). Unlike conventional CBA, which requires all benefits and costs to be expressed in monetary terms and the estimation of a mono-criterion such as net present values (NPVs) or internal rates of return (IRRs), MCA uses multiple criteria and both quantitative and qualitative indicators. Values attached to different aspects of biodiversity and ecosystem services by stakeholders can be analysed using MCA to arrive at scores on different aspects of value. This gives a relative ranking to alternate notions of value, and provides a crucial input into stakeholders' perceptions with respect to biodiversity and ecosystem services. However, a disadvantage of MCA is that when conflicting evaluation criteria are considered, an MCA problem may be mathematically ill-defined, making a complete axiomization and hence a simple decision criteria difficult to arrive at. Deliberative processes are considered to be a more inclusive method of decision-making through consultation with locals and stakeholders. However, in highly stratified and traditional agrarian societies, the voices of the poor and marginalized sections may go unheard while those of the rich and powerful interest groups may dominate.

Stakeholder approach to valuation is also recognized as a useful tool in order to solve social conflicts (Pascual et al., 2011). Different stakeholders attach different values to ecosystem services depending on their socio-economic and cultural backgrounds, and how these services impact on their livelihoods. A stakeholder approach can also help in evaluating who wins and who loses when possible management strategies are implemented in a socio-ecological system. An effective policy should address fairness and equity concerns (inter- and intra-generational), good governance, and ecological and institutional contexts (Pascual et al., 2011). A number of policy instruments such as payment for ecosystem/environmental services (PES), environmentally harmful subsidy (EHS) reforms, eco-certification, biodiversity or ecosystem services offsets, direct investment in natural capital, etc., have been implemented with a view to reward and support good conservation practices. The TEEB has documented examples of linking valuation with policy implementation from around the world (TEEB, 2011).

ABOUT THIS BOOK

This book addresses the methodological issues and challenges in valuing ecosystem services. It discusses different valuation tools as well as the importance of spatial heterogeneity in valuing ecosystem services. Thereafter it presents case studies that value ecosystem services covering a cross section of ecosystems such as forest, agricultural, coastal, coral reef and urban ecosystems in different sites, countries and regions from around the world. Finally, the book presents case studies linking valuation to policy and discusses the potential and experience with policy instruments such as PES schemes. The contributors to this book are leading experts from around the world who have made a significant contribution to the literature in this area. For convenience, the discussion in the book is organized under three parts: I Methodological issues and challenges; II Case studies: valuation; and III Case studies: valuation and policy.

PART I METHODOLOGICAL ISSUES AND CHALLENGES

Bateman et al. (Chapter 2) provide an overview of the key challenges and issues in integrating economic analyses with ecosystem services assessments. Focusing upon analyses for future-orientated policy and decision-making, they propose a general framework and nomenclature for undertaking such analyses under alternate scenarios. They initially consider a single period during which ecological stocks are maintained at sustainable levels. The flow of ecosystems services and their contribution to welfare-bearing goods is considered, and methods for valuing resultant benefits are reviewed and illustrated via a case study of land-use change. They then broaden their time horizon to discuss the treatment of future costs and benefits. Finally, they relax the sustainability assumption and consider economic approaches to the incorporation of depleting ecological assets with a particular focus upon stocks which exhibit thresholds below which restoration is compromised. They emphasize that substantial gaps in knowledge and data in the natural sciences and economics, other concerns such as discounting, economic valuations being based on current preferences and distribution of income which has implications for optimal social well-being, uncertainties regarding the impact of depleting ecological stocks, threshold effects and climate change on the provision of ecosystem services, etc., pose challenges in valuing ecosystem services.

There are several valid ways to value ecosystem services, and benefit transfer is one of them. However, straining to use existing point estimate values in the literature well beyond what they were intended for in the original study may undermine the credibility of all efforts to value ecosystem services. Such 'incredible' numbers may very well result in sinking the whole concept of ecosystem services, taking with it a valuable tool for informing public policy on use of natural environments. Loomis et al. (Chapter 3) suggest that using advances in benefit transfer methods developed in the last decade can increase the chances of estimating credible values for ecosystem services. These advances in methods are reviewed, and sources provided for new databases and new computerized models for valuing ecosystem services.

CVM is used to value habitats, species and ecosystem services. However, CVM surveys should be properly formulated and conducted in view of its limitations. One such problem is hypothetical bias. However, recent advances in research have found ways of tackling this problem. In Chapter 4, Mohammed presents the results of a study in Thailand which tries to examine the effectiveness of the widely used hypothetical bias mitigation techniques namely: follow up certainty question and cheaptalk, and a newly introduced ex-post mitigation technique called 'pledging'. His analysis demonstrates that certainty question and pledging could effectively reduce hypothetical bias, while cheaptalk did not have a statistically significant effect. In addition, further explanation is provided on factors that affect the likelihood of pledging.

Morse-Jones et al. (Chapter 5) report the results of a choice experiment survey that investigated the preferences of UK residents for the conservation of threatened wildlife in a biodiversity hotspot in Tanzania. They examine the sensitivity of values to species types, number of species and conservation sites, and to potential substitutes/complements. Critically, they find some evidence of coherency in preferences. Respondents are willing to pay significant amounts to conserve charismatic/endemic species and are scope sensitive to the number of endemic species. In contrast, species that are neither endemic nor charismatic, and the number of conservation sites, do not contribute significantly to utility. Further, changing the overall scope of the 'good' is found to have a significant and differential impact on respondent's choices depending on the species type: as the availability of wildlife increases, the authors observe substitution effects for non-endemic charismatic species, and complementarity for endemic (non-charismatic) species.

Spatial heterogeneity will impact on the provision and valuation of ecosystem services. Economic valuation of the ecosystem services provided by natural capital is complicated because these benefits manifest through interactions with human, social and built capital. Sutton (Chapter 6) explores several ways in which spatiality presents issues with respect to the valuation of ecosystem services, taking storm-protection services of coastal wetlands as a case study. In addition, a review of some recent research demonstrates why making spatially explicit evaluations of ecosystem services informs the utility and validity of these assessments, and how spatially explicit assessments can characterize which nations of the world are net ecosystem service providers or consumers.

Pollination services provide a variety of direct and indirect benefits to humans, including facilitating the production of food and feed crops, as well as contributing to overall ecosystem resilience. Declines in pollinator populations have raised concerns regarding potential risks to global food security and spurred efforts to quantify the benefits of pollination services. Studies focused on the economic valuation of pollination services have typically used one of three major approaches: (1) estimation of changes to social welfare, (2) calculation of the value of crop production that can be directly attributed to animal-mediated pollination, and (3) summation of replacement costs, whereby purchased inputs substitute for natural pollination services. Bauer (Chapter 7) compares the strengths and weaknesses of these valuation approaches across theoretical and empirical grounds. She suggests that while approaches that deliver measures of social welfare are preferable, no method is best for all situations and each method has some drawbacks including issues of cost and data availability.

Undeveloped and ostensibly unused land may be valuable for the services its natural ecosystems provide, such as pollination services. However, there is considerable uncertainty regarding the value of such services. In Chapter 8, Simpson considers pollination services that have attracted considerable attention. Local farmers can promote the growth and survival of insects, birds and other pollinators by retaining more of their lands in natural cover. Simpson develops a simple and stylized but illustrative model of economic choices in setting aside areas for pollinators. The model shows that a sort of Catch-22 is likely to prevail. Economic value is determined by scarcity. Pollinators are scarce – and, hence, valuable – only to the extent that otherwise harvestable crops are being lost for want of pollination. If crops are lost for want of pollination, farmers are likely to turn to alternative land uses, such as growing crops that do not require animal pollination or using their land for housing or industry. Thus an appeal to the value of pollinating services is unlikely to motivate farmers to set aside more of their land to provide ecosystem services. Rather an appeal to the broader concerns for preserving nature may be more effective in motivating conservation.

PART II CASE STUDIES: VALUATION

The chapters in this part present case studies involving the valuation of ecosystem services of different ecosystems. It also includes chapters that review forest valuation studies and coastal recreational benefits.

Ninan and Inoue (Chapter 9) provide a comprehensive review of studies that have tried to estimate the value of forest ecosystem services across forest sites in different countries and regions around the world. Evidence suggests that not only the total valuation of ecosystem services varies widely across studies but also the valuation of individual services. Their analysis suggests that policies to conserve ecosystems and their services should emphasize local contexts and values. They also discuss the shortcomings of existing studies, and suggest that, among others, future research should focus on the neglected ecosystem services, 'disservices', assess the role of dynamic factors and environmental catastrophes on provision of ecosystem services, and the benefits of keeping forests intact versus converting them to alternative uses.

Valuing ecosystem commodities, including ecological services, and determining their implications for the optimal management of ecosystems is challenging. In Chapter 10, Tisdell considers the optimal spatial use of forest ecosystems. The problem considered is whether it is optimal to partition the use of a forest so that a portion of it is used exclusively for wildlife conservation with the remainder being used for heavy logging (a dominant-use strategy) or to combine wildlife conservation and selective logging in at least part of the forest (a multiple-use strategy) with any remainder of the forest being available for heavy logging. The assumed objective is to maximize the profit from logging subject to the population of a focal forest wildlife species being sustained at a level that is at least equal to its minimum viable population. The optimal use strategy cannot be determined a priori but requires alternatives forgone to be assessed. While orangutans are used as an example, the model can be applied to other species or other ecological services such as the quality of water flowing from forested areas.

Forest ecosystems provide several intangible benefits which policymakers ignore since these values do not register in conventional markets or are difficult to measure. Drawing on results of a case study of a forest reserve in Japan, Ninan and Inoue (Chapter 11) suggest that the annual value of the estimated seven ecosystem services provided by the forest is not only worth billions of dollars, but also much more than hitherto known. They estimate this value for the Oku Aizu forest reserve to range at US$ 1.427–1.482 billion or about US$17 016–17 671 per ha. They suggest that if these are accounted for then governments and societies faced with the development versus conservation dilemma can make more informed decisions and policies that will help conserve forests and the ecosystem services they provide.

One of the drawbacks of existing forest valuation studies is the dearth of studies that have tried to assess the impact of dynamic factors and environmental catastrophes on the provision and value of ecosystem services. Beukering et al. (Chapter 12) estimate the non-market benefits provided by the forests in Montserrat in Trinidad and Tobago in the aftermath of a volcanic eruption. After the destructive impact of the volcanic activity starting in the mid-1990s, the Centre Hills now comprises the largest intact forest area remaining on Montserrat, providing a number of important ecosystem services to the people. Their study aimed at increasing understanding of the economic importance of further conservation of the area. The study covered a wide range of issues, addressed numerous ecosystem services and conducted two main analyses: a choice experiment (CE) and a CBA. The CE survey tried to elicit Montserrat population's value preferences for different attributes, such as local recreation, aesthetic quality, species abundance and control of invasive species (a growing threat to the forest ecosystem that has exacerbated after the volcanic eruption). The survey revealed that each household is willing to pay US$5 per month to control invasive species. Although the eradication and control of invasive pigs and rats in the Centre Hills generates substantial benefits over time, the extended CBA demonstrates that net benefits of an eradication programme would be negative, highlighting the need to lower the financial costs by developing cheaper eradication techniques.

Unlike forest ecosystems, the ecosystem services provided by agricultural and rural landscapes have received less attention in the literature. Yoshida (Chapter 13) has evaluated eight ecosystem services from agricultural and rural landscapes in Japan using a replacement cost method (RCM). The total economic value of these services was estimated at US$76.4 billion nationally, and US$33.7 billion for hilly and mountainous areas. The estimated monetary value per hectare was US$16 000. These results were used to devise a direct payment programme for conserving ecosystem services in hilly and mountainous areas in Japan. The study emphasizes the need to examine the role of land-use changes on the provision and valuation of ecosystem services from agricultural landscapes and also compare these with other alternate land uses so as to provide the economic justification for conserving agricultural and rural landscapes.

There is still a limited understanding about the role of nature in cities, and the approaches cities are using to enable ecosystem structure and function to improve the quality of life for urban residents. With the majority of the human population now living in urban areas, we need a better understanding of the natural processes that enable urban habitation. In Chapter 14, Shandas et al. use ecosystem services as a mechanism for linking biophysical and social systems, and describe a suite of factors that affect the availability of urban ecosystem services. By examining Portland (OR, USA) as their case study, the authors focus on three dimensions of cities that affect the types of services available: green building, vegetation and storm-water management. Where available they provide quantifiable measures of how these dimensions contribute to the economic, social and ecological quality of the study area. They recognize the need for further research in the area of urban ecosystem services, and highlight a set of promising tools for quantifying ecosystem services within and around urban areas.

Coastal ecosystems provide a variety of services such as recreational benefits. In Chapter 15, Ghermandi and Nunes examine the welfare dimension of these benefits. First, they construct a state-of-the-art database of primary valuation studies that focus on recreational benefits. They then develop a meta-analytical value transfer framework of the economic value of coastal recreation, which relies on Geographic Information Systems (GIS) for the characterization of the spatial context of the valued ecosystems. Context characteristics such as local climate, biodiversity richness, accessibility and anthropogenic pressure are found to significantly affect recreation values. Finally, the estimated econometric model is used to transfer and scale up the economic values for 368 regions of the European coastal zone. The authors analyse the distribution of aggregated regional and national recreation values and their composition in terms of the relative role of international, domestic and local recreationists.

Oyster reefs have lost an estimated 85 per cent of their historic extent globally. This loss carries a high economic cost because of the immense benefits they provide to humans. Recent research suggests that large-scale reef restoration is feasible, holding the prospect of concomitant recovery of ecosystem services and associated economic benefits. Kroeger and Guannel (Chapter 16) assess the biophysical flows and economic value of two key ecosystem services oyster reefs provide – fishery enhancement and coastal erosion reduction – for two reefs in Mobile Bay (AL, USA) currently being restored. They conservatively estimate that the 5850 m of reefs will yield 3100 kg of finfish and crab and 4855 kg of oyster meat harvests annually, generating annual mean social net benefits of US$87000. The reefs also reduce wave height and energy to levels that promote sediment accretion along the currently eroding shorelines. Reef restoration cost is below the average cost of conventional coastal armouring found along developed bay shorelines. Substituting reefs for conventional armouring along future shoreline developments could deliver an estimated 50-year social return on investment of 1.95 from fishery enhancement and avoided armouring cost alone – or US$2 in net benefits per dollar invested in reef restoration – with a net present value of US$1.11 million per mile of reef.

PART III CASE STUDIES: VALUATION AND POLICY

Valuation of ecosystem services is undertaken to improve decision-making and policies to conserve and manage ecosystems. This part presents case studies linking valuation to policy.

Coastal habitats are under increasing threat from human activities resulting in the loss of their critical ecological functions. Since the economic value of habitat loss is not easily captured in markets, financially connecting these resources to the roles they play in the global carbon cycle and the climate system is a way to curtail further losses. Appropriate financial incentives may be provided by carbon markets. Chapter 17 by Pendleton et al. analyses the extent to which the amount and value of carbon stored in coastal habitats can provide the financial incentives necessary to protect them. Their results show that gross returns for avoided mangrove conversion projects are the highest of the three habitats modelled. Also, the revenue potential from carbon credits outweighs the total costs of habitat protection for all habitats assuming a reasonable carbon price of US$15 per tCO_2e.

How many additional trees would a payment for a tree-planting project yield in a given location? Jindal and Kerr in Chapter 18 address this question amongst rural households in western Kenya. Based on a survey of 277 households the study follows an attribute-based method to elicit farmers' preferences. Demand is measured in terms of the additional number of trees that a household is willing to plant under different price schedules, including a direct, conditional economic incentive to plant new seedlings. The mean willingness to plant new trees per household increases from 44 trees when farmers have to pay Ksh10/seedling (about US$0.13 at the time of the study) to 244 trees when farmers receive a payment of Ksh10/seedling, conditional on survival. Their findings show that a relatively small incentive can yield a relatively large number of additional trees, particularly for timber species and for male respondents with plenty of household labour.

A significant share of global ecosystem service production occurs on privately held land, implying that efforts to sustain and enhance ecosystem services would benefit from a focus on private land managers. In Chapter 19, Kramer et al. examine ecosystem service markets as a possible mechanism for attaining conservation objectives on private lands. They report the results of a mail survey that assessed the attitudes of private farm operators in North Carolina, USA, towards conservation and ecosystem service programmes. The choice experiment survey included an estimation of tradeoffs across alternative structures of PES programme. While restricted in geographic scope, the results of the study may help in the design and success of PES programmes elsewhere. These include: (1) PES-type programmes, particularly those focussed on water quality and wildlife habitat, were of great interest to local farm operators. (2) Payment levels are an important factor in decisions to enrol, but so are other programme attributes, particularly contract length. (3) Campaigns seeking to raise awareness and/or enrolment in conservation payment programmes in general will benefit from a better understanding of preferred programme characteristics among potential participants. (4) Campaigns seeking to raise awareness and/or participation in PES programmes will meet with greater success if they can effectively focus on reaching land operators exhibiting particular characteristics.

REFERENCES

Boyd, J. and S. Banzhaf (2007), 'What are ecosystem services? The need for standardised environment accounting units', *Ecological Economics*, **63**, 616–629.

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

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton, M. van den Belt (1997), 'The value of the world's ecosystems services and natural capital', *Nature*, **387**, 15 May, 253–260.

Fisher, B., I. Bateman and R.K. Turner (2011), 'Valuing ecosystem services: benefits, values, space and time', Working Paper Series No.3, Ecosystem Services Economics, The United Nations Environment Programme, Nairobi.

de Groot, R.S., L. Brander, S. van der Ploeg, R. Costanza, F. Bernard, L.C. Braat, M. Christie et al. (2012), 'Global estimates of the value of ecosystems and their services in monetary terms', *Ecosystem Services*, **1**, 50–61.

Loomis, J. (2013), 'Strategies for overcoming hypothetical bias in stated preference surveys', paper presented at the 2013 Western Agricultural Economics Association Meeting, Monterey, CA.

Loomis, J., P. Kent, L. Strange, K. Fausch and A. Covich (2000), 'Measuring the total economic value of restoring ecosystem services in an impaired river basin: Results from a contingent valuation survey', *Ecological Economics*, **33**, 103–117.

MEA (Millennium Ecosystem Assessment) (2005), *Ecosystems and Human Well-being*: *Biodiversity Synthesis*. Washington, DC: World Resources Institute.

Nahuelhual, L., P. Donoso, A. Lara, D. Nunez, C. Oyarzun and E. Neira (2007), 'Valuing ecosystem services of Chilean temperate rainforests', *Environment, Development and Sustainability*, **9**, 481–499.

Ninan, K.N. (2011), 'Introduction: biodiversity, ecosystem services and human well-being', in K.N. Ninan (ed.), *Conserving and Valuing Ecosystem Services and Biodiversity*: *Economic*, *Institutional and Social Challenges*. London and Washington, DC: Earthscan, Chapter 1.

Ninan, K.N. and M. Inoue (2013), 'Valuing forest ecosystem services: what we know and what we don't', *Ecological Economics*, **93**, 137–149.

Norgaard, R. (2010), 'Ecosystem services: from eye-opening metaphor to complexity blunder', *Ecological Economics*, **69**, 1219–1227.
Ojea, E., J. Martin-Ortega and A. Chiabai (2012), 'Defining and classifying ecosystem services for economic valuation: the case of forest water services', *Environmental Science and Policy*, **19–20**, 1–15.
Pascual, U., R. Muradian, L. Brander, E. Gómez-Baggethun, B. Martin-Lopez, M. Verma et al. (2011), 'The economics of valuing ecosystem services and biodiversity', in P. ten Brink (ed.), *The Economics of Ecosystems and Biodiversity in National and International Policy Making*. London and Washington DC: Earthscan, pp. 183–255.
Turner, R.K., J. Paavola, P. Cooper, S. Farber, V. Jessamy and S. Georgiou (2003), 'Valuing nature: Lessons learned and future research directions', *Ecological Economics*, **46**, 493–510.
Vié, J.-C., C. Hilton-Taylor and S.N. Stuart (eds) (2009), *Wildlife in a Changing World*: *An Analysis of the 2008 IUCN Red List of Threatened Species*. Gland, Switzerland: IUCN.
WWF and ZSL (2012), 'Living planet report 2012: biodiversity, bio-capacity and better choices', World Wildlife Fund, Zoological Society of London, Global Footprint Network, European Space Agency.